

José Marrugo Negrete · M. P. Jonathan ·
AL. Ramanathan · S. Chidambaram ·
R. Nagarajan · Pankaj Kumar *Editors*

Mining Impacts and Their Environmental Problems

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Editors

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 Springer

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Preface

Mining activities is one of the critical drivers or pressures triggering triple planetary crisis and land degradation. However, the development of civilization along with science and technology makes mining as an inevitable process. Much of the infrastructure supporting modern societies can be attributed to mineral extraction and processing. Reports indicate that the process of mining leads to the environmental pollution, change in land use and land cover along with greenhouse gas emissions. This book depicts a holistic view of several global environmental issues due to impact of mining and also focusses on its solutions and challenges. With case studies across global south, different chapters in the book brings the readers across various aspects of mining sectors such as—(a) traditional and modern practices in mining; (b) pollution status due to mining activities (soil and water); (c) risk exposure and vulnerability to human health; (d) state of art technologies being used across different regions to inform decision and policy makers to help them design robust management plans in order to minimize environmental degradation and sustainable transition in the future.

First section spanning from Chaps. 1–4 focussing on the theme “environmental pollution due to mining practices” highlighted pollution in sediments and water, GIS application to assess the impact of open pit mining; water quality and pollution index; impact of Nickel mining on freshwater fishes and their impacts in food chain; impact of coal mining, gold and chromite mining on water resources and soil in the adjoining regions. The studies were representative of cases in Latin America, Colombia, Sub-Saharan region, and India.

Next section spanning from Chaps. 5–7 focusses on “Mining hazards and reclamation of abandoned mines” focuses on risks and challenges to environment and human health/health risk in Sub Saharan Africa and India due to mining activities. The detailed study on mining impact in Uttarakhand hills ecology brings out the effect on the pristine environment due to the mining activities in the region. Further studies on modelling focussing the health risk exposure to mercury (Hg), brings out the probability of health impacts and novel studies on the treatment of gold mine tailings and advanced oxidation process for cyanidation in wastewater treatment proving the effectiveness of treatment methods. Another research on the status and extent on

the impact of mining on water resources in a northern state of India indicates that groundwater has reached an alarming stage of contamination and necessary steps are to be taken to prevent further contamination in the region. Next section covered by Chaps. 8–10 discusses an array of topics related to carbon capture, neutralization and reclamation relevant for contemporary policy issues at global landscape. One section highlights the role and possibilities of using mining waste for carbon storage and sequestration and the key factors controlling this removal process are its mineralogy, particle sizes, and grain size. Furthermore, it discusses the importance of advanced oxidation technologies, biotechnical technologies and its feasibility in grinding the mining waste, which improves the CO₂ removal rate, reduces the land requirement for mining waste disposal, reduced operation and maintenance cost; all required for scaling it up at the industrial level.

Last section spanning from Chaps. 11–13 revolves around the role of environmental policy and governance for managing various issues from mining sector. First this section discusses about how to reconcile the commercialization of jewellery products that maintains an economic dynamic in Mexico with the minimization of environmental impacts due to the extraction and use of silver for the generation of monetary income derived from commercial activity. This study raises the importance of proper management of silver production to establish positive signals for a less carbon-intensive transition. It is followed by a review summary for decarbonization policies in Mexico and how various initiatives putting its endeavour toward transition to a low-carbon economy for the year 2050 that has been agreed upon by the Sustainable Development Solutions Network (SDSN) of the United Nations. These initiatives range from heavy investment in institutional reforms, stakeholders' engagements, participatory approach etc. to accommodating all the necessary elements for sound implementation of decarbonization policies. Finally, this section discusses some of the effects of mining on coastal ecosystems and some of the alternatives to mitigate them. For a long time, coastal ecosystems have remained relatively untouched by the carbon offset market, however, in recent years their great capacity to sequester GHG has begun to be recognized. Although there are few methodologies applied to coastal ecosystems, there is enormous potentiality to use existing ones and even develop new ones. The conservation of these areas will not only allow the capture of GHG but will prevent their degradation from releasing these gases into the atmosphere. From an environmental perspective, taking advantage of marine ecosystems is more sustainable than other type of projects.

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Ciudad de México, Mexico
New Delhi, India
Safat, Kuwait
Miri, Malaysia
Hayama, Japan

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Chapter 1

Mining Tailings and Dam Failures in Latin American Mining Countries



A. F. Zúñiga-Cabezas, P. F. Rodríguez-Espinosa, and S. S. Morales García

1 Introduction

Nature and natural resources are inextricably linked to societal progress in Latin America. Nature–society relationships provide both opportunities and problems in creating more fair, egalitarian and sustainable development in this resource-rich region. The region has over half of the world’s tropical forests and various other natural biomes; it has a quarter of the world’s potentially arable land and one-third of the world’s freshwater reserves that combinedly contain a wealth of biodiversity. Despite five centuries of extractive activities to meet global demand, the Latin American region still has significant mineral deposits, including oil, gas, iron, copper and gold. On the other hand, as a result of the spread of extractive activities and land-use changes, this biodiversity hotspot has suffered rapid loss of biodiversity, increasing ecosystem degradation, and one-third of the world’s carbon emissions (Baud et al., 2011).

Mining generated approximately \$604,830 million between 2010 and 2012 for the Latin American countries (Economic Commission for Latin America and the Caribbean [ECLAC], 2013). Mining accounts for up to 10% of the gross domestic product (GDP) of Mexico, Bolivia and Colombia, and more than 15% for Chile (Aguilar-Pesantes et al., 2021). However, despite the wealth it provides, some Latin American public still has reservations about new mining projects because of the long history of mining-related issues and environmental and technological catastrophes

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that have occurred in the region, just as the August 2014 Buenavista del Cobre disaster in Cananea, Sonora, Mexico, where the Buenavista del Cobre mine spilled 40,000 m³ of copper sulfate acid and high levels of arsenic into public waterways near Cananea, causing damage to the Tinaja, Sonora and Bacanuchi rivers and uprooting 25,000 people in seven counties along the river (Toscana Aparicio and Canales Hernández, 2017). Mexico's Secretary of Environment and Natural Resources called it 'the worst environmental disaster by the mining industry in modern times', citing hazardous toxins as a significant threat to human health and a dramatic loss of public drinking water (Schwarz-Herion and Omran, 2020). The incident caused health problems for at least 360 persons, 322 wells were forced to close, and farmers in the affected areas were not able to sell their products because of buyer concerns about contamination. The disaster directly affected 24,000 people and indirectly impinged on 250,000 people in seven settlements along the Sonora river, 25 miles south of the Arizona border, with the Federal Attorney General's Office for Environmental Protection (PROFEPA) estimating that the accident could have exceeded \$134 million in environmental damage (Schwarz-Herion and Omran, 2020).

Similarly, the San Marco dam disaster, a mining accident in Brazil on November the 5th 2015, which took place at the Fundão dam near Mariana City, Minas Gerais State, affected the Doce river (the second-longest river in Brazil). The river was submerged in a 'sea' of red mud as a result of the mud wave caused by a tailings dam failure, which was estimated to have spilled 43 million m³ of tailings and extended across several kilometers, travelling 665 km, eventually reaching the Atlantic Ocean, with a significant negative impact on marine ecosystems and coastal surroundings (Hatje et al., 2017; Silva Rotta et al., 2020). For many nearby settlements, this river served as their main source of water and fish. Along with the broad and significant water pollution, the tragedy also claimed the lives of 20 people. It further resulted in the mass extinction of fish, inundated an area covering the Bento Rodrigues district of about 7,000 km², and damaged the Abrolhos Marine National Park, with the annual loss of environmental services to be roughly US\$521 million (Garcia et al., 2017).

2 Contamination in the Mining Process

Modern society could not function without the products of the extractive mining industries, which are vital components in, for example, airplanes, ceramics, computers, construction materials, metals, and paint. On a global scale, the mining industry also provides direct employment to over 40 million people, along with indirect support for approximately 200–250 million people (Kossoff et al., 2014). According to Venter et al. (2016), between 1993 and 2009, the global economy grew by 153%, while the human population climbed by 23%; this exemplifying a rapid increase in resource use because of the quicker economic expansion.

The potential for water, air and soil contamination that mining represents, along with the grievous landscape transformation that most modern mining entails, make environmental issues dominant among rationale for critiquing contemporary mining

demowith environmental problems emerging at almost every stage of the mining process, with some of the most common being the following:

- **Exploration stage:** Construction of access roads, mapping, drilling of trenches and boreholes, sampling, establishment of waste dumps, and occasionally blasting are all activities done during the process of mining. These operations have been frequently linked to detrimental environmental effects, such as excessive logging of vegetation during the clearance of the zone, degraded soil and drilling fluid and waste rock, and contamination of water and soil resources (Asr et al., 2019; Lottermoser, 2007). The use of explosives results in morphological damage to structures, the stressing of wildlife and cattle, and respiratory issues from the dust (Helwege, 2015).
- **Exploitation and mineral-processing stage:** Depending on their scale, exploitation and processing techniques, mining operations have a variety of negative environmental effects. The destruction of natural landscapes and ecosystems, as well as mine falling and land subsidence, water, soil, air, noise, thermal, and radiation pollution, the exposure of sulfide-rich tailings to precipitation causes acid mine drainage (AMD) with the heavy metals that AMD transports, failure of the tailings dam, explosion, and spontaneous combustion are some of the most significant environmental effects of these stages (Asr et al., 2019). Excessive use of water, destruction or disturbance of natural habitats or sites of cultural significance, and the accidental or deliberate release of solid, liquid or gaseous contaminants into the surrounding ecosystems are other damaging environmental effects of mining operations (Lottermoser, 2007).

3 Tailings Dams

Mineral processing of hard rock metal ores (e.g. gold [Au], copper [Cu], lead [Pb], zinc [Zn] and uranium [U]) and industrial mineral deposits (e.g. phosphate and bauxite) involves size reduction and separation of individual minerals. In metal mining, the extracted ore minerals represent only a small fraction of the whole ore mass; the vast majority of the mined material ends up as tailings. More than 99% of the original material mined may finally become tailings in case low-grade metal ores are utilized (Lottermoser, 2007). For example, a good-grade copper deposit may contain only one-quarter of 1% copper; therefore, it involves grinding (milling) the ore and separating the relatively small quantities of the metal from the waste material, which results in very fine particles that not only allow more efficient extraction of metals but also generate a high volume of waste materials called tailings (Veltmeyer, 2016). To protect the environment from their harmful properties, these fine suspended materials, known as tailings, are typically kept as slurry in sizable man-made embankments, known as tailings dams (Islam and Murakami, 2021; Lyu et al., 2019). The global mining sector has created enormous amounts of tailings, with an estimated 14 billion tonnes exclusively produced in 2010 (Owen et al., 2020; Shui-Quan et al., 2022). The current trend of declining ore grades suggests that mining

companies will require to store and manage a significant amount of future tailings (Kossoff et al., 2014).

The long-term goal of tailings disposition and management is to prevent the mobilization of toxic materials and their release into the environment, often stored in impoundments behind dams, which can fail, with ensuing environmental, human health and economic consequences (Kossoff et al., 2014). These tailings dams should be built to indefinitely keep waste materials and should be designed to: (a) provide long-term stability against erosion and mass movement; (b) prevent environmental contamination of ground and surface waters; and (c) restore the area for the future land use. Ensuring a secure and stable tailings dams after operation should be the main goal of tailings dam designing (Lottermoser, 2007).

4 Tailings Dam Failures

Failures of tailings dams have caused terrible disasters for both environment and people, as found by Piciullo et al. (2022). According to the various tailings dam databases reviewed, a total of 257 failures have been documented since 1915, resulting in about 2,650 fatalities and 250 million m³ of toxic residues released into the environment, with approximately 640 fatalities and 50% (115 million m³) of this released volumes documented after 2000. These statistics demonstrate how the volume and complexity of the problem of safely storing mining waste are increasing, as some of the threats are made more difficult by the increased frequency of climate changes and the severity of extreme weather occurrences.

Currently, information on tailings dam failures is commonly obtained from databases, such as those of the International Commission on Large Dams (ICOLD), the United Nations Environment Programme (UNEP) and the World Energy Information Service (WISE), Centre for Science in Public Participation (CSP) and World Mine Tailings Failures, with several authors also making compilations of information on tailings dam failures, for instance Guimarães et al. (2022), Islam and Murakami (2021), Kossoff et al. (2014), Lyu et al. (2019), Piciullo et al. (2022), Rana et al. (2021, Table 1); Rico et al. (2008), Shui-Quan et al. (2022) and Zhang et al. (2009). However, one common conclusion in these reviews is that the information about dam rupture incidents is inconsistent, and although these published reviews are valuable, they are incomplete because smaller instances are frequently reported insufficiently in both scientific journals and by the public. In addition, many, if not the majority, of failures are believed to go unreported because of worries about negative publicity and legal repercussions, especially in China and Russia. (Kossoff et al., 2014).

In one of these compilations, Shui-Quan et al. (2022) categorized 342 failures of tailings dams globally, with every 25 years as separate phases, with the years 1947–2021 (the years 1915–1946 were removed due to data loss) being divided into three phases: 73 tailings dam collapses during the first phase (1947–1971), 143 tailings dam collapses during the second phase (1972–1996), and 115 tailings dam failures during the third phase (1997–2021). The yearly average of global tailings

Table 1 Major tailings dam failures worldwide (2000–2020)*

<i>Site</i>	<i>Country</i>	<i>Year</i>	<i>Site</i>	<i>Country</i>	<i>Year</i>
Aitik	Sweden	2000	Obed Mtn.	Canada	2013
Baia Mare	Romania	2000	Dan River	USA	2014
Inez	USA	2000	Mt. Polley	Canada	2014
Sasa	Macedonia	2003	Fundão	Brazil	2015
Comurhex	France	2004	Luoyang	China	2016
Cavendish	UK	2007	Jharsuguda	India	2017
Mineracao	Brazil	2007	Mishor Rotem Power Station	Israel	2017
Kingston	USA	2008	Tonglvshan	China	2017
Xiangfen	China	2008	Cadia	Australia	2018
Karamken	Russia	2009	Cieneguita	Mexico	2018
Ajka	Hungary	2010	Cobriza	Peru	2019
Caudalosa	Peru	2010	Essar	India	2019
Las Palmas	Chile	2010	Feijão	Brazil	2019
Kayakari	Japan	2011	Hindalco	India	2019
Gullbridge	Canada	2012	Luming	China	2020
Philex	Philippines	2012	Singrauli	India	2020
Talvivaara	Finland	2012			

* Adapted from Rana et al. (2021).

pond failures was at least 4.4 during 1947–2021, 5.7 during 1972–1996, 4.6 during 1997–2021 and 2.9 during 1947–1971. The most frequent tailings dam disasters occurred between 1972 and 1996 (Cavalheiro Paulelli et al., 2022). Dam collapses increased in South America and Asia in the twenty-first century, with 35 dam rupture events between 2010 and 2020. In South and North America, eight instances of dam failures each were reported, with a total of 16 failures; 50% of the incidents in North America were located in Canada, and 62.5% of the incidents in South America were caused in Brazil (Guimarães et al., 2022).

4.1 Reasons for Tailings Dam Failure

The tailings dam failures have a variety of reasons, and the corresponding patterns and mechanisms are quite complex; failures of tailings dams are frequently the consequences of various causes acting in concert rather than being caused by a single factor (Shui-Quan et al., 2022). In general, the causes of ruptures are not clear in the literature and when noted, causal factors should be understood just as an indication of the triggering aspects of the events, as the information presented often failed to

determine causal factors unequivocally, and several catastrophes are still under investigation. Their categorization is a complex task, as their reference in the literature may be based on different criteria and methodologies that often point to the same mechanism causing the catastrophe but in different ways (Reis et al., 2020).

Following the analysis of multiple reviews of tailings dam failures (Adamo et al., 2020; Guimarães et al., 2022; Islam and Murakami, 2021; Kossoff et al., 2014; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Rico et al., 2008; Shui-Quan et al., 2022; Zhang et al., 2009), books and technical reports (Lottermoser, 2008; US Environmental Protection Agency, Office of Solid Waste, 1994), and databases of tailings dam failures and based on the classification of causes for tailing dam given by ICOLD, some of the most frequently named factors are as follows:

- **Structure and condition of the dam foundation (FN):** The stability of the dam body may be harmed by foundations that have experienced significant pressure and horizontal thrust because of deformation or slide. The two key contributing factors to this phenomenon are: (1) weak foundation of the dam (especially if karst foundation is present), and (2) the base of the dam has poor slip resistance. The slip resistance of the foundation may decrease due to foundation fracture, rock softening or alterations (Adamo et al., 2020; Islam and Murakami, 2021; Kossoff et al., 2014; Lottermoser, 2008; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Reis et al., 2020; Rico et al., 2008; Shui-Quan et al., 2022; US Environmental Protection Agency, Office of Solid Waste, 1994; Zhang et al., 2009).
- **Earthquakes (EQ):** The weakening of tailings dam components, destabilization of the dam itself, and liquefaction of tailings sand are the three methods through which earthquakes result in tailings dam failure. Furthermore, the primary elements influencing the dam's seismic liquefaction are the dynamic load circumstances, the quality of the physical characteristics of the tailings, and burial conditions (Adamo et al., 2020; Islam and Murakami, 2021; Kossoff et al., 2014; Lottermoser, 2008; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Rico et al., 2008; Shui-Quan et al., 2022; US Environmental Protection Agency, Office of Solid Waste, 1994; Zhang et al., 2009).
- **Mine subsidence (MS):** In general, this occurs if the land is unable to handle the intense strain caused by tailings pond settlement or collapse, which results in dam failure. Typically, these tailings ponds are constructed above an underground mine (Adamo et al., 2020; Islam and Murakami, 2021; Kossoff et al., 2014; Lyu et al., 2019; Piciullo et al., 2022; Rico et al., 2008; Shui-Quan et al., 2022).
- **Structural inadequacies, inadequate or failed decants (ST):** This happens if there are errors in design or a if component' fails to perform as intended, for instance unsuccessful decants used to remove water from impoundments (the most frequent component to fail) (Adamo et al., 2020; Islam and Murakami, 2021; Kossoff et al., 2014; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Reis et al., 2020; Shui-Quan et al., 2022; Zhang et al., 2009).
- **External erosion (ER):** In a structure, erosion by water flow can be classified into two types: external erosion, when water erodes the visible portion of the structure,

and internal erosion, when water erodes an invisible element of the building or its base. Large-scale rainfall can erode the soil and wash it away through runoff, possibly leading to this factor of tailings dam failures (Adamo et al., 2020; Islam and Murakami, 2021; Lottermoser, 2008; Lyu et al., 2019; Piciullo et al., 2022; Shui-Quan et al., 2022; US Environmental Protection Agency, Office of Solid Waste, 1994).

- **Seepage and internal erosion (SE):** The material of tailings may be deformed by infiltration under the influence of seepage; when the infiltration deformation requirements are reached for tailings dam, a pipe surge happens in dam's body, creating cracks and local pond collapse as well as local instability of the dam slope, ultimately resulting in the failure of the tailings dam (Adamo et al., 2020; Islam and Murakami, 2021; Kossoff et al., 2014; Lottermoser, 2008; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Reis et al., 2020; Rico et al., 2008; Shui-Quan et al., 2022; U.S. Environmental Protection Agency, Office of Solid Waste, 1994; Zhang et al., 2009).
- **Overtopping (OT):** Failure of dams caused by overtopping (flooding) is summed up as follows: water may impact the dam or the water level may rise, resulting in the eventual failure of the dam due to high wind or rainfall, drainage infrastructure blockage or destruction, and diminished flood discharge capacity. The main cause of flooding in tailings ponds is typically rainfall, with impermeable dams, outdated flood relief infrastructure, scale obstruction, damage, and other phenomena acting as secondary causes (Adamo et al., 2020; Islam and Murakami, 2021; Kossoff et al., 2014; Lottermoser, 2008; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Reis et al., 2020; Rico et al., 2008; Shui-Quan et al., 2022; U.S. Environmental Protection Agency, Office of Solid Waste, 1994; Zhang et al., 2009).
- **Slope instability (SI):** This refers to the damage brought on by the building or extrusion of tailings sand, significant rainfall, or other circumstances (non-seepage disturbance). The dam body slip resistance or cracks in the dam are typically not enough to cause the tailings dam failure, so it's usually accompanied by other factors. Cracking and bulging of the dam face, protrusion of the slope, sinking of the top of the dam, and distortion of the dike border are typical symptoms of degradation (Islam and Murakami, 2021; Kossoff et al., 2014; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Reis et al., 2020; Rico et al., 2008; Shui-Quan et al., 2022; U.S. Environmental Protection Agency, Office of Solid Waste, 1994).

In a study of 342 dams around the world, Shui-Quan et al. (2022) found the causes of 258 failures (causes of the other 84 failures are still unknown), with the known causes being: 10.1% by ST (26), 22.1% by SI (57), 17.1% by EQ (44), 5% by ER (13), 11.2% by SE (29), more than 24.4% by OT (63), 0.4% by MS (1), and 9.7% by FN (25), with most of the tailings dam failures directly related to heavy rainfall or earthquakes, which consented with the data reported by others studies (Islam and Murakami, 2021; Kossoff et al., 2014; Piciullo et al., 2022; Rana et al., 2021; Reis et al., 2020; Rico et al., 2008). These refer either to *weather hazards* (which

include heavy rainfall, snowfall, storms and freeze-thaw cycles) or earthquakes at the top of the reasons for dams failures. Weather hazards may lead to several failure mechanisms, such as overtopping, internal erosion (piping), seepage, excess pore pressures, and structural drainage issues, which may become an even bigger problem, given the fact that tailings dam failures triggered by weather hazards have steadily increased in frequency over time. This is consistent with the findings of Rico et al. (2008), Azam and Li (2010) and Lyu et al. (2019) about tailings dam failures in general; while earthquakes could induce liquefaction failures or seismic tremors that accelerate the impoundment towards failure (Rana et al., 2021).

The available data also suggests that active (rather than inactive) and upstream dams are most likely to fail; this was concluded from a thorough review of European tailings dam failure data (Rico et al., 2008), which observed that 83% of failures occurred when the dam was active, 15% occurred in inactive and abandoned dams, and only 2% occurred in inactive but maintained dams, with Kossoff et al. (2014) and Piciullo et al. (2022) also supporting the claim that active tailings dams have a bigger chance of failing. Nevertheless, the inactive tailings dams (a dam that has been filled or one where the mine production has ceased) also represent a problem, on their own, being that as soon as a tailings dam becomes inactive, it continues to be maintained usually only in the cases where the current environmental legislation states the obligation of supervising and controlling inactive dams (as in European countries); however, in all other cases, especially in countries without a strong environmental legislation (as in some countries of Latin America, Africa and Asia), the majority of tailings dams are abandoned, lacking any type of monitoring or control (Rico et al., 2008). It is likely that a significant contributory factor accounting for the comparative stability of inactive tailings impoundments lies in the ongoing oxidation and cementation of tailings in an unsaturated zone, estimating that cementation may increase the liquefaction resistance of the impoundment by as much as 250% over 30 years (Kossoff et al., 2014). In addition, cementation may have the advantage of sequestering potentially harmful substances, such as lead and arsenic, in later phases, for instance those found in calcite, gypsum and jarosite-group minerals (Jia and Demopoulos, 2005; Smith et al., 2006).

The data also shows that a significant percentage of failed tailings dams have been those that were raised in the upstream direction (Franks et al., 2021; Kossoff et al., 2014; Lyu et al., 2019; Piciullo et al., 2022; Rana et al., 2021; Rico et al., 2008). Given the expensiveness to construct and maintain tailings dams, mining corporations frequently choose less expensive solutions for the dam's construction, within three broad categories: upstream, downstream, and centerline dams (although recent hybrid designs are also available) (Roche et al., 2017). In general, the upstream type of dams are the cheapest option, as they are typically constructed from the rock materials available onsite and require fewer construction materials (Islam and Murakami, 2021; Lyu et al., 2019); however such dams have been known as 'unforgiving structures' (Martin and McRoberts, 1999), for which good design practice and stewardship are vitally important to maintain their integrity.

The failure of an upstream dam because of static liquefaction (defined as the material losing its solid qualities in reaction to applied stress, which causes the material to

behave more like a liquid) is one of the main possible risks, with this behaviour being very harmful to the stability of tailings dams. In seismically vulnerable places, the applied stress may occur from mine blasting or the movement of heavy equipment, but earthquakes pose the greatest threat (Kossoff et al., 2014), with some authors even concluding that upstream raising is not a suitable construction method in seismically susceptible areas (U.S. Environmental Protection Agency, Office of Solid Waste, 1994) or others strongly emphasizing the risks involved; stressing the importance of strong designs and good maintenance practices precisely customized for the site in question, taking probable failure modes into consideration (Kossoff et al., 2014).

5 Overview of Tailings Dams Failures Distribution

The rate of tailings dam failures over the past century has been widely reported as roughly 1.2% annually (Adamo et al., 2020; Azam and Li, 2010; Islam and Murakami, 2021; Lyu et al., 2019; Shui-Quan et al., 2022); however, this information only covers a portion of all dam failures caused by incomplete and erroneous data, as the majority of incidents were not reported, especially for the period before the 2010s, as tailings dam incidents might not be reported in some developing countries with lax environmental regulations or government-controlled public media (which could explain the higher number of failures in developed countries with strict environmental laws, such as the United States and the European region) (Islam and Murakami, 2021), with more recent studies suggesting that the actual annual rate is closer to 3.45% (Islam and Murakami, 2021) or 4.4% (Shui-Quan et al., 2022).

It is also worth mentioning that the failure rate of tailings dams is significantly higher than that of other conventional water-retaining dams (Adamo et al., 2020; Azam and Li, 2010; International Commission on Large Dams, 2001; Islam and Murakami, 2021; Kossoff et al., 2014; Lyu et al., 2019; Shui-Quan et al., 2022), with the main reasons for this higher susceptibility being: (1) embankments constructed with soil, coarse waste and residual materials from mining operations; (2) the amount of wastewater increases with increase in the height of tailings dams; (3) lack of reasonable regulations on design standards; and (4) the cost of monitoring a tailings dam is high during mine operation and closure (Rico et al., 2008).

5.1 Geographic Distribution

Concerning the geographic distribution of failure of tailings dams, a recent study conducted by Shui-Quan et al. (2022), in which the causes and regional distribution patterns of 342 tailings dam failures globally from 1915 to 2021 were reviewed (Fig. 1), found that 21.3% of tailings dam failures occurred in Asia, 12.6% in Europe, 57.9% in the Americas (with the biggest participation for North America), 4.8% in

Africa, 2.6% in Oceania, and 0.8% in unknown regions, with other authors establishing similar trends, with North America and Asia at the top of the list in the regional distribution of dam failures (Islam and Murakami, 2021; Lyu et al., 2019; Rico et al., 2008; Zhang et al., 2009). On the other hand, for the temporal distribution of tailings dam failures (Islam and Murakami, 2021) in a study of 366 incidents compiled from the ICOLD and WISE databases spanning from 1915 to 2020 (with the first reported incident in Chile in 1915) discovered that the number of tailings dam failures remained low until the 1950s, after which a sharp rise in the number of failures is noticeable (which may have been caused by the growing demand for metals globally), with a total of 235 events between 1960 and 1999 (accounting for approximately 64% of all instances) and another rising tendency in the 2010s, in spite of a lower number of recorded events in the 2000s. Also, the same dataset when categorized into two periods (1910–1999 and 2000–2020) clearly demonstrated a change in the distribution pattern of tailings dam failures, with a significant decline in both North American and European failure instances, with North America at 79% of the instances before 2000 but only at 21% thereafter, as there were much fewer occurrences in the U.S. mines. Other nations, such as Canada (19 cases before 2000 and 11 cases after 2000), Chile (32 cases before 2000 and nine cases after 2000), and the United Kingdom (15 cases before 2000 and just one case after 2000) also saw the same pattern. Recent events, however, indicate that the trajectory of tailings dam breaches has changed from industrialized to developing countries, with prominent failures reported in Brazilian, Chinese, and Mexican mines between 2000 and 2020 making this tendency clear. In addition, when the data on the production of mines was classified by continents, an intriguing pattern emerged: after 2000, the production of mines in Africa and South America almost doubled, but it was around seven times lower in Europe. In contrast, production of mines with incidents in Asia and North America decreased after 2000, although not significantly (Islam and Murakami, 2021) (Fig. 1).

For the Latin American region, some of the most important tailings dam failures in recent history are as follows:

- About 32 million m³ of tailings gushed out and flooded 158 homes in villages downstream from the iron ore waste dam owned by Minas Gerais Cérrego do Feijó Vale at the Brumadinho city of Brazil. The dam (that had been operating at a high-water level for years) failed briefly on the 25th of January, 2019 as a result of unstable dam slopes, leaving 259 people dead and 11 missing and polluting 650 km of waterways. This has been regarded as the most serious mine dam failure in history (Silva Rotta et al., 2020; Thompson et al., 2020).
- At least 17 people were killed and 650 km of the Doce river that flows into the Atlantic Ocean was contaminated when the Samarco iron ore tailings pond in Fundão, Brazil collapsed on the 5th of November, 2015 because of a small earthquake that caused dam liquefaction. The failure released more than 50 million m³ of mining tailings with an estimated cost of approximately U.S. \$20 billion for recovering the impacted ecosystems. This environmental catastrophe happened over a distance of 500 km until it finally reached the Atlantic Ocean (Demajorovic

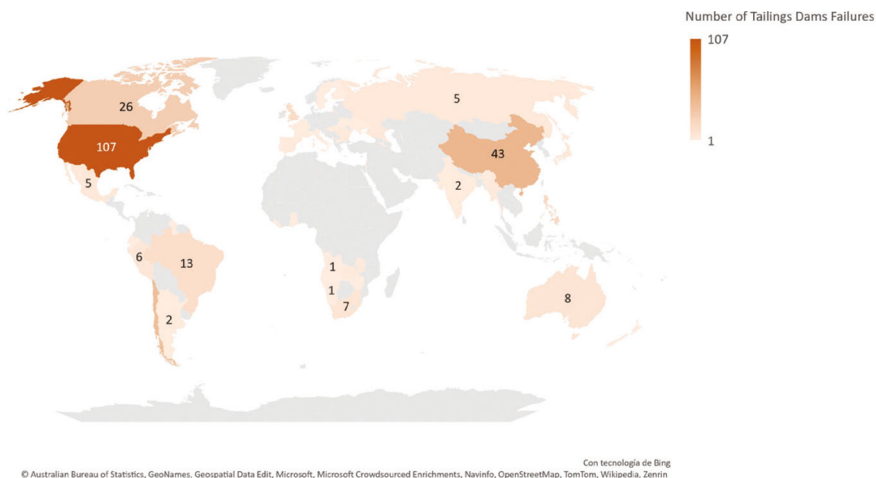


Figure 1: Global geographic distribution of tailings pond dam failures. (Source: Shui-Quan et al., 2022)

et al., 2019; dos Santos Vergilio et al., 2021; Garcia et al., 2017; Gomes et al., 2017; Hatje et al., 2017).

- In August 2014, 40,000 m³ of sulphuric acid poured into the Sonora river in the State of Sonora (Mexico) because of a structural defect in the primary mining tailings storage dam that could not resist the impact of unusually heavy rainfall. Over 420 km of the river's ecosystem was affected by the tailings (Aguilar-Pesantes et al., 2021; Toscana-Aparicio and Hernández-Canales, 2017).

6 Influence of Dam Breaks on Legislation

Based on the reports prepared by Guimarães et al. (2022), 285 ruptures occurred in the twentieth century, 8.8% dated before 1950 and 91.2% occurring from 1960 to 2020. Triggered by an increase in the number of ruptures, this in 1990 prompted the United Nations Environment Program in partnership with the International Council on Metals and the Environment to organize meetings with diverse stakeholders to explore the risk associated with the disposal of tailings, construction of dams, failure occurrences, environmental effects, and risk management.

In 1999, the World Business Council on Sustainable Development created the Global Mining Initiative, which developed the Mining Minerals and Sustainable Project Development (MMSD). The report prepared by MMSD emphasized that tailings management is a challenge to be solved by encouraging better handling practices, rather than simple disposals in dams. Nevertheless, its last update on the report occurred in 2002 (Owen et al., 2020) prior to failure of several dams, such as the 2008 Taoshi Dam failure in China, the 2014 Mount Polley Dam failure in

Canada, and failures in the Fundão (2015) and Brumadinho (2019) dams, both in Brazil (Guimarães et al., 2022).

Other significant normative steps were taken in 2006, when the European Union implemented Directive 2006/21/EC, which emphasizes the importance of mining waste management. In 2015, at the 3rd Conference on Disaster Risk Reduction, the United Nations set a target of significantly lowering disaster risks and their effects by 2030, discussing subjects, such as better resource management, information generation and sharing, prevention tactics, and rehabilitation measures (Guimarães et al., 2022).

In addition, based on the recent events, such as the catastrophic tailings dam failures in Mariana (2015) and in Brumadinho (2019) in Brazil, large entities, such as the Mining Association of Canada (MAC) and the Federal Emergency Management Agency of the United States (Federal Emergency Management Agency, 2019), between others, elaborated a wide range of guidelines, standards and regulatory documents. All these now available to assist an operator for establishing a tailings risk assessment and management system, and emergency action plans, such as the Risk-Informed Decision-Making (RIDM) in the ISO 2394:2015 (International Organization for Standardization, 2015) and the *Guide to Tailings Facility Management* produced by MAC in 2017 (its third edition was released in 2019; MAC, 2019). All management systems and action plans were established to improve the safety management of tailings dams and provide a management framework, policies, objectives, and checklists to be implemented throughout the lifecycle of a tailings dam (MAC, 2019). FEMA's (2019) *Dam Incident Planning Guide* was released to set parameters for the emergency response plan. The International Council on Mining & Metals (ICMM), the UN Environment Program and the Principles for Responsible Investment (PRI) also issued the *Global Industry Standard on Tailings Management* (GISTM; ICMM et al., 2020) to set out the requirements for emergency response plans in scenarios of tailings dam failures (Guimarães et al., 2022; Piciullo et al., 2022).

6.1 Latin American Region

As mentioned before, the trajectory of the tailings dam failures has shifted from developed nations to the developing world. This shift brings some major questions regarding the capacities of developing nations to tackle the consequences associated with tailings dam failures (Islam and Murakami, 2021). In developed countries (such as Canada), following the Mount Polley disaster in 2014, Mount Polley Mining Corporation acted swiftly and removed most of the spilled tailings in Lower Hazeltine Creek within a few months of the accident. This strategy undoubtedly reduced the short- to long-term chemical impacts in the watershed (Byrne et al., 2018). In contrast, the consequences of the Fundão (2015) and Brumadinho (2019) tailings dams failure in Brazil are ongoing and in litigation, although some corrective measures had been already implemented (Silva Rotta et al., 2020; Thompson et al., 2020). The History

and evolution of some of the legislative and regulatory efforts by some of the countries most affected by tailings dam failures are shown below.

6.1.1 Brazil

Regarding the mining industry, the Federal Law N° 227 (1967), which assigned responsibility to the nation for the management of mineral resources, mineral production, distribution, trade, and consumption, was the first legal and national regulation in Brazil. However, this document had a gap regarding the extractive industries proper environmental obligations (Guimarães et al., 2022). Then in 1999, the São Paulo branch of the Brazilian Committee on Dams published a *Basic Guide on Dam Safety* based on the Canadian Dam Safety Guidelines, accessible to dam owners and operators as a general reference (Morgenstern, 2018; Scaletta et al., 2012).

In 2010, the National Policy on Safety of Dams and the National System on Safety of Dams in Brazil were established to address the Federal's lack of a policy for tailings dams (Presidência da República, 2010). The goals of this law were to ensure dam safety requirements fulfillment, control of dam safety requirements during various stages of the dam project, encouraging monitoring and oversight, implementing public involvement, establishing technical guidance, and promoting a culture of dam safety and risk management (Morgenstern, 2018).

Then, after the failure of the Fundão (2015) and Brumadinho (2019) dams tailings in the state of Minas Gerais, Brazil, the international community critically evaluated the situation, scrutinizing the issue of global tailings dam failures and its unprecedented impact (Islam and Murakami, 2021); resulting in numerous laws being published and updated. The Norm ABNTNBR13028:2006 (which could be considered as the first regulatory instrument explicit for tailings storage facilities) was updated by a designed committee, releasing its new version in 2017, ABNTNBR13028:2017. The updated version elaborated considerably on technical requirements to support approval of the design of a tailings dam and described relevant international practices in this regard (Morgenstern, 2018). In addition, new standards were set for environmental licensing in the state of Minas Gerais, Brazil. New standards included aspects about the communication procedure in the event of dam accidents, evacuation plans for people who are in flooded areas, and survey of damaged structures with a consensus on all updates on the value of action plans that ensure the supply of drinking water in the event of a dam failure (Guimarães et al., 2022).

6.1.2 Chile

In Chile, the development of a regulatory framework related to tailings deposits began after a 7.5 Richter earthquake struck Chile's central region, collapsing most of the region's tailings deposits, and causing destruction and deaths in the communities affected. This event sparked an instant response at both national and global levels,

sparking discussions regarding the creation of a regulatory framework for tailings deposits and to address the safety of mine tailings and technical advancements of these projects. On 31 July 1970, these symposiums led to the issuance of Decree No. 86 of the Ministry of Mining (which was in effect until December 2006). This legislation regulated the construction of these deposits and prohibited the upstream construction method, as this was deemed as being more susceptible to the seismic nature of the country (Ghorbani and Kuan, 2017; Ministerio de Minería Gobierno de Chile, 2019; Morgenstern, 2018; Rana et al., 2021).

Without limiting the above since the 1994 implementation of the General Bases of the Environment Law (Law N° 19.300), which established the System of Environmental Impact Assessment (SEIA) and its corresponding Regulation in 1997, mining projects have been required to plan for and assess their environmental impacts. Furthermore, in 2004, under the provisions of D.S. No. 132, Mining Safety Regulations of the Ministry of Mining, whose purpose was to protect the life and physical integrity of people working in the mining industry, regulations on the closure of mining sites were included for the first time. The regulations mandated mining companies to execute a closure plan at the end of their productive phase, including the closure of tailings deposits (article 495) (Ministerio de Minería Gobierno de Chile, 2019).

Subsequently in 2007, the Ministry of Mining published D.S. No. 248, Regulation for the Approval of Tailings Deposit Design, Construction, Operation and Closure Projects of tailings deposits. With this regulation repealing the D.S. No. 86/1970, established new guidelines for the approval procedure of mining tailings deposit projects, and included requirements for the design, construction, operation and closure of the deposits with the goal to ensure safety of people (Ghorbani and Kuan, 2017; Ministerio de Minería Gobierno de Chile, 2019).

Lastly, to reduce the negative effects of the mining sector and prevent the abandonment of sites after the suspension of their operations, in 2011, the Ministry of Mining published Law No. 20,551, which governs the closure of mining sites and facilities, differing from the prior regulations. It mandates the performance of a risk analysis of the facilities' long-term behaviour, the proposal of measures to control the risks of effects on the environment and human safety, and the assessment of all works, measures, and closure activities to subsequently provide financial guarantees to ensure that the State has the financial resources to carry out an efficient and effective closure of tailings deposition facilities (Ministerio de Minería Gobierno de Chile, 2019).

6.1.3 Mexico

The most important historical period that influenced the orientation of the regulation of mining activity was the 1990s. During this period, the country began a process of commercial opening, undertaking for this purpose several constitutional reforms (Article 27) as well as the drafting of a new Mining Law (1992). This new Mining Law is the legal order that regulates the use of existing mineral resources through the

granting of concessions to individuals for exploration, extraction, and benefit, and followed the neoliberal interest of the time, with objectives, such as facilitating the undertaking of projects, restricting participation by national companies, protecting foreign direct investment (FDI), decreasing the requirements for requesting exploration and exploitation concessions as well as limiting state intervention for the creation of new taxes in the industry (Azamar Alonso, 2018; Secretaría de Medio Ambiente, 2004)

The constitutional reform to Article 1 in 2011 indicates that, within the framework of current regulations, the laws (local, federal and international) that favour individuals over companies are the ones that have greater weight in any legal dispute, and have allowed legislation, such as the International Labor Organization (ILO) Convention 169 (also known as the Indigenous and Tribal Peoples Convention, 1989), to be used in favour of the affected communities. Finally, in 2014, a mining reform facilitated the creation of some taxes that are more focused on actual practice, as tax obligations are established for precious and rare metals. However, the exploitation of other metallic and non-metallic minerals, stock market speculation and rent from concessioned hectares are still not regulated (Azamar Alonso, 2018).

Concession holders are also obliged by the mining law to take care of the environment and ecological protection under the legislation and regulations on the matter during all stages of the mining process (exploration, exploitation and beneficiation) and to comply with the general provisions and with the Mexican Official Standards (NOM) applicable to the mining-metallurgical industry regarding safety, ecological balance and environmental protection of tailings dams. In Mexico, the main norm is the NOM-141-SEMARNAT-2003 (Secretaría de Medio Ambiente, 2004), which establishes the procedure for characterizing tailings as well as the specifications and criteria for site preparation, design, construction, operation, and post-operation of tailings dams. The norm also lays down specific factors, such as site selection and investigation, initial pear, unsatisfactory foundations, lack of downstream slope stability, among others, to be considered for tailings dams.

7 New Trends in Tailings Dams Management

In management of tailings dams, in order to minimize tailings volumes and improve dam safety, it is a common practice to dewater tailings to a paste-like consistency. However, dewatering further to a bulk cake like consistency has been typically used for modest operations in the areas where seismicity or topography makes dam construction impossible. In addition to the trend towards socially responsible investment, recent high-profile failures of tailings dams at plants, such as Fundão and Brumadinho, have reignited interest in the dewatering of tailings (Furnell et al., 2022).

7.1 Dewatering

One of the most obvious aspects of the global mining sector is improper tailings management practices. Severe and long-lasting environmental and social impacts could be the result of tailings spills, dam failures, seepage, unrehabilitated sites, and incidents of direct discharge into waterways with the potential to have long-lasting negative impacts on the environment, society and economy. The mining of lower-grade ore and the resulting rise in waste production per unit of resource, along with competition for water and other resources, have the potential to exacerbate tailings management's current difficulties (Mulligan et al., 2011).

Responsible tailings deposition techniques should make sure the tailings are: (1) physically, chemically, and radiologically stable; (2) inert if they are to interact with the environment; (3) if they are not inert, geographically bound and managed with consideration of environmental and social conditions at each location; and (4) managed to minimize energy and water consumption. In this sense, insufficient water management could result in instability of the tailings dams in several ways, such as partial saturation of the dry structural area (which can cause a static failure), inadequate water drainage, excess of tailings that exceeds the dam's design parameters, and overloading a dam with erodible structural material because of the tendency in tailings dams management to reduce the amount of water stored (Furnell et al., 2022).

In general, ore deposits located in Latin America, that is in countries with dry climates (such as Chile and Mexico) where fresh water is used to handle and transport tailings hydraulically to tailings dams, given its more affordable nature than bulk transportation through conveyor belts, trains, or trucks, water supply may become a major challenge, making an effective water management strategy more important than ever to manage tailings throughout the mine's operating life (Cacciuttolo and Valenzuela, 2022). In addition, as communities, agriculture, and other productive sectors require more water, several factors (including climate change) have an increasing impact on water resources, which in turn causes the water supply, recharge, and infiltration flows in basins to drop gradually; water resources have become more vulnerable, resulting in a conflict between the demands of various water users. Consequently, the design and operation of tailings dams are constantly being improved with an emphasis on the creation of optimal solutions for water management, with a main focus on factors such as technology dependability, a dynamic and robust tailings dam water balance (taking into account site-specific conditions), and prevention of water loss (evaporation, infiltration, and retention in interstitial voids) to obtain reduced water requirements, less adverse environmental effects and increased access to natural water sources for the community (Cacciuttolo and Valenzuela, 2022).

7.2 *Emerging Technologies in Dewatering*

Since dewatering to >70 wt% solids is very energy and capital-intensive, the advantages of innovative dewatering technologies cannot be disregarded (Furnell et al., 2022). The development of dewatering technologies has been the subject of extensive research, and some potential possibilities are as follows:

- ***Volute screw press:*** The auger screw-based volute screw press, also known as a volute dewatering press, dewater material as it is forced into a drum. The auger screw rotates while pushing mobile rings in the drum, which is made up of alternating fixed and eccentric movable rings that glide past one another. The movement of the moving rings between the fixed rings keeps the interior of the screw press clean and helps to break up material clumps to release more water. At the screw's tip, a dewatered cake is forced out while filtrate escapes via the gaps between the rings that make up the barrel (Furnell et al., 2022).
- ***Superabsorbent polymers (SAPs):*** SAPs are hydrophilic, cross-linked polymer networks consisting of ionic monomers that have a large water or aqueous solution absorption and retention capacity. The low degree of cross-linking allows them to absorb up to 1,000 times their weight (Mignon et al., 2019). SAPs are made from synthetic monomers, such as acrylates or acrylamides, or from natural polymers, such as polypeptides and polysaccharides. Water is physically trapped by capillary pressure in the SAP macroporous structure, where it is then hydrated by SAP functional groups to cause absorption (Furnell et al., 2022; Mignon et al., 2019).
- ***Electrokinetic dewatering:*** The cations in an aqueous suspension flow towards the cathodes when an electric field is applied across it, together with bound water and surrounding free water, due to viscous forces. Accelerating electro-osmosis and electrophoresis involves passing a DC current through geotextiles sandwiching saturated soils or via a filter press (Furnell et al., 2022; Vaxelaire et al., 2010).
- ***Geotextile tubes:*** Geotextile tubes filled with bulk sand or silt, although created for coastline protection, have been researched for use in coal residue storage, sewage sludge removal, and dewatering of mining tailings. The filtrate can be collected by setting up and stacking geotextile tubes loaded with slurries on an impervious pad. There is also research on reagent addition to this process to enhance the quality of the filtrate and assist solids consolidation, in a similar way to other dewatering procedures. This technology offers benefits such as the total encapsulation of wastes in geotextile tubes, effectively shielding them from fugitive dusting in windy situations, having a small environmental impact, holding solids without harming the ecosystem, and efficiently carrying this waste to disposal facilities (Furnell et al., 2022; Kiffle et al., 2023).

8 Conclusions

In spite of increasing advances in legislations and protocols for dam risk management, adequate design and diligent operation, and closure of mining sites in the Latin American region, these legislations still lack stronger regulations to ensure a continuous and effective monitoring and evaluation of tailings dams, as even up to the standards and regulations of dams. Given the complexity of factors that affects dam's stability, regulations could fail due to the presence of unexpectedly seismic events, unusual weather phenomena, or even the naturally variable quality of a construction project that could span for several decades. In this sense, it is important that the government and communities alike in countries with relevant mining industries push for stricter laws for the monitoring of tailings dams by the state and institutions as well as by the society. It is the obligation of mining companies to fully share information related to the development and state of their tailings dams. Recent attempts by different researchers and organizations to compile in databases information related to the failures in the operation and status of tailings dams have been relatively successful. Several studies have already pointed out the lack of public disclosure in cases of tailings dam failures and the risks associated with these failures, given the reluctance of mining companies to share information related to their tailings dam facilities, even after major failures in the face of an ever-growing presence of mining activities in the Latin American countries. Safety and legislative restrictions in these countries are not in the forefront of operators' mind, but the necessity of diligent information for reducing potential failures may bring overwhelming results.

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Chapter 2

Geographical Information Systems (GIS), a Tool for the Assessment of the Impact of Open-Pit Mining in Exploitation Areas in Colombia



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1 Introduction

Mining investment provides great economic benefits for the countries that exploit such resources, especially those in development, where the elaboration of political frameworks can turn them into an important focus for foreign investment. Considering this, in the last decade, efforts have increased to establish adequate regulations for the mining industry, especially due to the opportunities for development and economic growth that relate to them (Himley, 2019). However, mining activities represent a series of almost irreversible negative impacts at the ecosystem level and on the benefits that these natural systems provide us (Wunder, 2005). In addition to this, the monitoring of said impacts has become a challenge due to the lack of innovative methodologies, the illegality, and the difficulty in obtaining data due to the inaccessibility of certain areas (Rajesh, 2004), including those of the Colombian territory.

Open-pit mining is an unavoidable and unsustainable activity if adequate monitoring of the exploited areas is not achieved (Padró et al., 2019). Currently, it has been described that this activity could be linked to the processes of deforestation, persistent pollution and crop losses, leading to the development of devastating social, environmental and economic conflicts in these areas (Schueler et al., 2011). Despite

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this context, mining continues to be important in maintaining the economic indices of many countries (Kivinen et al., 2018), without considering the great uncertainty related to the socio-environmental gaps that arise during the development of mining concessions (Kivinen et al., 2018), making monitoring and control processes difficult processes (Choi et al., 2020).

Latin American countries, especially the Amazonian countries, suffer the harmful consequences of mining, but above all the role of the illegality of this activity directly (Sonter et al., 2017). In the last two decades, illegal mining has progressively increased throughout the region due to the lack of adequate national regulations and the evident rise in international prices of raw materials, thus violating the socio-environmental rights of the communities that depend on it directly because of ecosystem services for their subsistence (SPDA, 2014). In the Colombian context, illegal mining covers between 50% and 80% of extractive activities, extending to approximately 65% of the territory. In this sense, the impacts of these activities range from the transformation of the territory to the depletion of the environmental supply and the deterioration of ecosystems, thus making the evaluation and formalisation of these processes increasingly urgent (Juárez, 2016).

The main stages of development of a mine are based on planning, construction, operation and recovery. Processes that are necessary to guarantee that the exploitation of a mining title is most appropriate for the territories, avoiding the presence of risks for both the environment and the man, which is why the development of technologies based on geographic information systems (GIS), within these phases, are often recommended for the optimisation of resources (Choi et al., 2020). From this point of view, it is necessary to determine on a spatiotemporal scale the impacts that mining would generate in order to understand its dynamics. Components that would likely be facilitated by both development of GIS and data that can be obtained through monitoring techniques, such as remote sensing systems (Maus et al., 2020), and technologies would certainly go hand in hand with sustainable development objectives (Craynon et al., 2016) by mitigating the unfavourable impact of mining.

From the perspective of the GIS, the processes associated with Artisanal and small-scale mining (ASM), an activity derived from open-pit mining, concentrate their greatest activities on geographic areas rich in natural resources but with low economic development (Green et al., 2019). In the case of Latin America, many of the mining processes are carried out under the framework of illegality, environmental degeneration, and a failure in the monitoring and planning processes of these activities (Sonter et al., 2017). In the case of Colombia, there are still information gaps on the capacity that GIS would have for the development of studies or methodologies that allow them to be taken as reference frameworks that enable their use in decision-making at the regional level (United Nations Office on Drugs and Crime and Government of Colombia [UNODC], 2020).

Geographic information systems are still considered a comprehensive framework used to query, store and analyse spatial data with various applications within the sciences (Zhou, 2021), being effective management support in decision-making for mine managers during the different phases of mining development (Choi et al., 2020). Satellite images, unmanned flights and GPS, among other technologies, are part of

those global positioning systems that have been useful for georeferencing territories for more than 50 years (Rajak et al., 2019). Thus, allowing to guide, control and automate mining processes remotely, being particularly valuable for territories where the operator's health is at risk or where obtaining information is difficult; provided more information on the processes that may be occurring in them. Under this overview, remote sensing based on remote sensors, and as seen in this chapter, could be considered for the evaluation of environmental impacts of these activities, being key to their monitoring (Turner et al., 2007). This could be very useful to generate a reference framework that could be used by state entities and decision-makers in the processes of mitigation, monitoring and control of this activity in territories considered vulnerable. In this context, the objective of this chapter is to contextualise and analyse how GIS, together with remote sensing systems (Remote Sensing), is beneficial as a tool to mitigate the direct impacts that have already been documented for mining, and to identify future directions of GIS in open-pit mining.

2 Remote Sensing Tools

Geographic information sciences have managed to benefit from the most recent technological advances, taking full advantage of both access to robust portable equipment and its low cost (Keil et al., 2021; Zhou, 2021). Globally, the development of GIS applications and software has succeeded in supporting the communication and sharing of spatial data. Recently, the development of technological resources based on remote sensing has been advanced, such as synthetic aperture radar imagery (RADARSAT), ASTER satellites, Sentinel satellites, Landsat satellites (Landsat Multispectral Scanner, Landsat Thematic Mapper and Landsat Enhanced Thematic Mapper) and its generation 1–9, SPOT satellites (System probatory observation de la Terre), Indian remote sensing (IRS) satellites (Rajesh, 2004), and even the use of satellites, such as Google satellite (as well as GIS and Google Earth), among others (EOX IT Services GmbH, 2018).

The different satellite technologies mentioned above have been able to provide detailed spatial and spectral information over time due to the properties of penetration and spatial resolution that they present on the earth's surface (Zhou, 2021; Sitjar and Busquets, 2015). However, despite the multiple characteristics that make them superior, their use in tropical latitudes with adverse atmospheric conditions can lead to both errors in data collection and their inaccessibility in some regions, mainly in the fields of mineral exploration (Sabins, 1983).

Another technology currently available consists of the use of unmanned aerial systems that have allowed the monitoring of mining processes with a spatial resolution of up to 10 cm. It is estimated that the efficiency in the flight times of these technologies allows the identification of an area greater than 10 hectares in an average period of 2 hours; that is why, the importance of this method for studies related to remote sensing in mining areas is highlighted (Padró et al., 2022). Additionally, this type of technology is equipped with hyperspectral sensors that allow the capturing

of information for multiple numbers of bands and essential inputs to improve the spectral monitoring of territories (Padró et al., 2022).

Similarly, other tools could be considered useful for obtaining data of mining importance, such as the use of Global Positioning System (GPS) (Zarubin et al., 2021). It is an American technology accessible since the 1970s for military purposes, which has been made available to civilian entities for navigability purposes (Rajak et al., 2019), being available to date on almost all smart portable devices. It is well known that the development of this technology has facilitated the monitoring, control and automation of mining processes globally, being a valuable resource mainly in areas where the health of the operator is at risk, for example, in underground mining (Li and Zhan, 2018).

2.1 Principles of Satellite Remote Sensing in Mining

Remote sensing refers to the acquisition of information remotely through the use of sensors (both passive and active, from planes, satellites, drones and GPS) that achieve the collection of spatial data. In practical terms, remote sensing is defined as the use and processing of images and data that have been acquired through remote sensing technologies which record the activity between matter and energy (Sabins, 1999), allowing the determination of its characteristics. Within the practical applications for the use of this technology (GIS), there are some sciences, such as meteorology, geology, geography and, lately, the health sciences (Davenhall and Kinabrew, 2012).

The use of remote sensors is based on the principles of electromagnetic radiation (Zhou, 2021) (depending on the frequency and wavelength that manage to disperse perpendicularly and continuously). Generally, these waves propagate through space, transporting energy from one place to another and with them the information about the objects that interact synergistically (spectral signatures), comprising ranges of wavelengths that go from kilometers to much smaller scales (Sitjar and Busquets, 2015). For mining, the use of remote sensing under the expend of spectral signatures is based on the principles of reflectance (amount of light reflected from an object), which are usually grouped as bands, by presenting similarities in the intrinsic properties of the elements that compose them (wavelength vs reflectance of the object) (Gaffey, 1985).

Thus, the movement of the vibrational transitions of the molecules, when monitored with the necessary technology to detect these variations, would produce anomalies in the reflectances that are specific to each element (Gaffey, 1985; Sitjar and Busquets, 2015). For example, the detection spectra for iron oxide minerals are usually identified in a range of 0.35 to 1.5 μm (Hunt and Ashley, 1979); consequently, any element that is over this range, by remote sensing, would be widely associated with deposits of this mineral. Gold, for example, has been commonly associated with the spectral identification of iron oxides, as well as quartz veins and geological fault processes (Rekhibi et al., 2015), for which its presence on territories could be linked to sites potential to carry out extraction processes successfully.

Making this technology under this concept is a useful tool in the development of the processes that reduce environmental impacts and increase the productivity of mining areas.

Currently, satellites have been developed with different spectral ranges (visible infrared, near-infrared, shortwave infrared, medium wave infrared, longwave infrared, and thermal infrared) as well as with an increase in the number of identified bands (1 to 11 on Landsat, for example) (Hubbard and Crowley, 2005). These technological advances have managed to be very useful for both lithological and structural mapping of minerals, as well as for the development of mineral exploration strategies, facilitating field work in situ and in real-time. However, some limitations must be considered for the use of this type of technology, such as the reduced number of available bands of some satellites and the standardisation of radiometric calibration techniques on the images obtained, among other processes related to spatial resolution (Sitjar and Busquets, 2015).

To solve this type of problem, the use of hyperspectral technologies has been launched recently through airborne systems and advanced satellites to improve the detection of minerals of industrial interest (information that is generally required to increase the classification of potential mining sites), thus allowing both automation of exploration processes and precision of absorption characteristics associated with their identification and the spectral resolution that composes them (Van der Meer and Bakker, 1997; Hubbard and Crowley, 2005). At the same time, the use of radar technologies (systems that actively provide their own power source on the ground) allows the penetration of their wavelengths through clouds and rain, managing to determine with greater precision the potential detection sites of minerals – this being the methodology used successfully in countries, such as Indonesia and New Guinea (Sabins, 1983) for some decades, and later implemented in the development of web platforms linked to open-pit mining in Colombia (Saavedra, 2022).

It is recognised that in order to improve the quality of the satellite images obtained, there must be a balance between spatial resolution, spectral resolution and radiometric precision (Denniss et al., 1994), for which the development of satellites or sophisticated remote technologies allow coming closer to meeting these assumptions (Hubbard and Crowley, 2005). For example, greater identification capabilities have been achieved for some minerals with the calibration of spectra through known ground targets (Hubbard and Crowley, 2005). Likewise, there is evidence of the benefits that the parallel use of different remote sensors would provide, for example, the ASTER satellites managed to establish a clear distinction between clays, sulphates and carbonates due to the greater number of bands (>200) available from their satellites. In the same sense, the results obtained from other airborne hyperspectral sensors, such as the aerial imaging spectrometer (AVIRIS) (Crowley et al., 2003), a high-cost technology for spectrum analysis, over time, would come to be supplemented with the improvement of atmospheric calibration techniques of the sensors that currently orbit the earth, mainly between the signal–noise ratio and the spatial–spectral resolution (Hubbard and Crowley, 2005).

3 Contextualisation of GIS in Mining

In this section, a brief overview of the importance and applicability that GIS has had along with the use of remote sensing technologies in recent years is made, particularly emphasizing their use for the development of mining studies, and briefly describing their results.

From a general perspective, the use of GIS for the estimation of mineral reserves under the implementation of geostatistical methods, such as Kriging precision (Uygucgil and Konul, 2015), has allowed estimating the shape, location and volume of mine of open-pit mining in countries such as Turkey, generating great advances in mining-planning processes. Similarly, Baek et al. (2016) proposed useful uncertainty quantification methods to improve, from another point of view, the optimisation of resources in mining. In this context, it would be prudent to mention other applications of GIS as a technology that has made it possible to better predict and plan mining dump areas based on available topographical and environmental information, evaluating the potential they would have on the impacted territories (Yenilmez et al., 2011).

Lechner et al. (2017) proposed the use of GIS as a feasible tool to facilitate the most appropriate positioning of machinery in the exploitation processes through least-cost route analysis and social survey methods for the identification of corridors with the least possible impact on the environmental values of a territory. From a social point of view, Jerónimo et al. (2015) were able to analyse the land use planning processes against social interests in Cajamarca, Peru, serving the GIS as a basis for conflict zones for the development of mining titles in an optimal way. Likewise, and in terms of operational efficiency, Dijkstra (1959) presented a method of available networks for an open-pit mine to optimise the travel period of trucks over the exploited territories, thus improving both operation of the mine and its impacts.

Geographic information systems, together with the use of remote sensors, both alone and in parallel, have proven to be useful tools for the management and mitigation of impacts in mining (Choi et al., 2020); this would lead to sustainable development and mutual benefits with natural resources (Lara-Rodríguez et al., 2020). Given this situation, Miao et al. (2008) managed to determine the reproducibility of the use of interferometry, using ERS (European Remote Sensing satellite) and ENVISAT ('Environmental Satellite') images for terrain monitoring compatible with mining activities, a technique based on the superimposition of light signals from different receptors to improve image resolution. On the other hand, Gorokhovich et al. (2003) and Kim et al. (2012) managed to combine the use of spatial data on natural and mining resources, extrapolating the GIS analysis for the prioritisation of feasible sites for rehabilitation and restoration processes, respectively.

From a hydrological perspective, the use of Hortonian analysis of drainage networks, a model based on geographic information on flow accumulations, has been used as a tool for the characterisation of impacted water bodies, allowing their order, distances and directions to be determined (Jaseiwicz and Metz, 2010). Variables

for which GIS would facilitate data collection through the creation of programming-based modules allow the generation of results through a minimal coding effort by the operator.

Determining the areas where mining processes are developed would allow us to measure the spatial scale to which this problem would be aiming. Maus et al. (2020) managed successfully to map 295 of the most important sites in terms of mining production around the world. The foregoing through the use of remote sensors with immediate access, for example, satellites, such as the Google satellite, Microsoft Bing Imagery and Sentinel-2 cloudless (EOX IT Services GmbH, 2018), enables the identification of mining areas greater than 57,277 km², with data precision greater than 97%. This being a starting point for the monitoring and temporal evolution of the main mining sites around the world, thus promoting training of data for both validation and automated classification of mines (Maus et al., 2020), such as monitoring and the social, environmental and economic impacts that have been generated on these territories.

4 Satellite Monitoring Systems as Management and Planning Tools

Mining continues to be a problematic activity for the territories where it is carried out (Lara-Rodríguez et al., 2020) due to the fact that it is carried out without a thorough evaluation of its environmental pressures (Pueblo, 2016), particularly when they are not directly related to reforestation activities that mitigate their impact (Argota, 2017). Illegal mining is one of the economic activities that are mostly permeated by organisations outside the law, making it difficult to use strategies that remedy the socio-environmental and economic consequences that relate to them (Lara-Rodríguez et al., 2020). Serving as an example of the foregoing, the regions of La Mojana and El Bajo Magdalena in Colombia lack the implementation of management plans for mining activities, in addition to the presence of social and economic conflicts that make it difficult to obtain monitoring data directly (Argota, 2017; UNODC, 2020).

It is estimated that a large part of the mining activities in Colombia is generated informally, leading to poverty and lack of development in the exploited territories. In this context, the processes of illegality generate strong difficulties to develop occupational health plans, deteriorating more and more the quality of life of the population (International Labour Office [ILO], 2015). For this reason, the use of technologies based on remote sensors is necessary, together with the use of GIS, since they facilitate the identification of illegal mining areas in order to generate measures that allow the formalisation, regulation and control of the same. To justify this importance, the case of the territory of Chocó in Colombia is taken as an example, a region with high mining potential that has historically been linked to the highest rate of poverty in Colombia. Therefore, the development of this type of studies would have a favourable impact because of the conditions that interconnect illegal mining

with the low socio-economic indices of the territory (Lara-Rodríguez et al., 2020; UNODC, 2020).

In both national and international context, the use of easily accessible technologies, in situ and in real-time, could be considered efficient inputs for decision-making in regions where exploration and monitoring is a necessary and complex task (Agudelo et al., 2012). In this sense, the use of remote sensors could be linked as an accessible tool to solve this problem, thus allowing the regulation, monitoring and surveillance of territories by systematically associating the information obtained with the GIS that is currently available in the country; such is the case of GIS for National Territorial Planning and Ordering – SIGOT (Gutiérrez and Urrego, 2011).

Generally, mineral resources tend to be circumscribed in small areas where the biogeochemical processes of the earth allow them to be concentrated above normal abundances. The probability of finding minerals varies according to some parameters, such as time, magnetic and gravimetric processes, biogeochemical processes and terrain elevation, among others (Rajesh, 2004). There are methods that in recent years have allowed us to identify minerals through different remote sensing techniques, including the use of images from satellites orbiting the earth, aircraft with spectral sensors, and unmanned flights, among others. Thus, the use of spectral analysis, such as the use of hyperspectral images (Sabins, 1999; Zhou, 2021), has become a useful tool to improve the mapping and detection of different minerals, based on absorption techniques and the use of specialised GIS software for its corresponding geospatial analysis. The use of this potential technology to provide information on the location of mineral deposits has now been recognised. However, it should be made clear that the use of these techniques does not seek to set aside the direct exploration of these areas; on the contrary, they allow to be additional techniques for mineral identification processes (Osinowo et al., 2021), thus leaving knowledge gaps that need to be strengthened to validate their results.

5 Some Remote Sensing Techniques for the Detection of Minerals

Within the techniques developed for mineral mapping, and as detailed below, there is the use of studies related to hydrothermal alterations (Osinowo et al., 2021), principal component analysis (Osinowo et al., 2021), band ratio composition, spectral indexes (Pour and Hashim, 2015) and Crosta analysis (Crosta and Moore, 1989), among others. Their use individually or in combination makes them appropriate tools for both georeferencing of minerals and input for the analysis of spatial data that facilitate decision-making and planning of mining concessions according to the availability and abundance of these resources.

5.1 *Principal Component Analysis (Pca) and Crosta Analysis*

Principal component analysis is a classic statistical method that generates images (components) under the linear combination of images that compose them. The application of PCA generates a relative variation of the observed images that have been successfully used to improve topographic information in structural mapping of minerals, being a favourable tool to minimise the effect of vegetation in remote sensing studies (Ruiz and Ledesma, 1998). Similarly, Crosta analysis techniques (a technique that determines the PCA that exhibits the greatest spectral contrast) have allowed PCA to be a variant for eigenvector analysis in typing principal components that contain spectral information of valuable interest (Crosta and Moore, 1989; Osinowo et al., 2021), mainly when it is related to the geological and structural mapping of minerals, thus improving the visual discrimination of areas with mining potential.

5.2 *Image Ratio and Band Composition*

It is a method that allows dividing the value of one band by another in order to highlight different values between bands that emit similar values, thus showing ranges of reflectances characteristic of a mineral. A variable of this technique, called band composition, allows the combination of three images for the red, green and blue (RGB) colour channels, manages to highlight characteristics of each element, recognising the multispectral classification of the image when assigning to each pixel, and classes according to its spectrum. This type of methodology can be classified by the operator, which specifies the classes to be used, or it can be non-supervised, for which the software assigns a random value (Sabins, 1999).

Based on the previously mentioned geolocation techniques, scientific evidence has validated the use of Landsat 8 satellite image data to map the geological characteristics and spatial distribution of hydrothermal alteration sites associated with gold mineralisation sites at Pala in southwestern Chad. Methodologies allow delineation of hydrothermal alteration zones based on ground observation in a more objective manner (Osinowo et al., 2020).

Similarly, the use of these methodologies facilitates the study of physicochemical phenomena of hydrothermal alterations, processes that significantly improve the contrast of mineral characteristics through qualitative identification (Ali and Pour, 2014). However, band ratio methods remain one of the best solutions for the detection of hydrothermally altered rocks, proving to be efficient for the identification of areas of high mineralisation power from the use of satellite data (Osinowo et al., 2020).

5.3 *Spectral Indexes*

Likewise, gold extraction sites have been identified from the generation of other methodologies based on spectral indices, such as normalized difference vegetation index (NDVI) and normalized difference water index (NDWI), through the use of Landsat images (Yang et al., 2018), facilitating the identification and classification of new mineral extraction areas in French Guiana (Gond and Brognoli, 2005; Yang et al., 2018). In this sense, GIS is still considered an important input in the estimation of restoration areas using different methodologies (Carabassa et al., 2021), minimising and monitoring their impacts on the environment (Zarubin et al., 2021).

Under this framework, it is possible to make a call to the scientific community for the development of mineral studies based on the methodologies described above for the Colombian territory due to the good results that have been obtained as well as its easy applicability and reproducibility. Taking into account that these require the use and development of increasingly sophisticated satellites or remote sensors that allow improvement in both spectral resolution obtained and depth of their wavelengths, obtaining more reliable terrestrial information in the areas of the Colombian territory with high cloudiness and with the presence of open-pit mining. These considerations have already been implemented through the studies carried out by Saavedra (2022).

6 **GIS as a Tool for the Evaluation of the Environmental Impact of Open-pit Mining**

Geographic information systems can generate spatial relationships with other data sources to combine or superimpose layers of information for later analysis (Bonham, 1994). The use of GIS, together with remote sensing tools, is often used in combination to extract-related information about a particular area of interest. The GIS in mining, as mentioned above, has made it possible to obtain the most approximate location of the concentration of minerals. Similarly, they have made it possible to determine and spatially distribute the behaviour of pollutants, such as heavy metals from a geographical point of view, facilitating the taking of measures to mitigate their impact through prospective studies for the identification of punctual emission zones (Cheng et al., 2020), mainly in the areas circumscribed to mining titles (Li et al., 2000, Lysaniuk et al., 2021), where they tend to be issued more frequently.

Scaling the applicability of GIS from an environmental perspective, these have not only facilitated the identification of mining areas through remote sensing but have also provided relevant information for determining changes in land cover near mining concessions (Flórez-Yepes et al., 2017; Ozelkan et al., 2012). Thus, these have allowed to establish a series of parameters, such as the sites where there are greater losses of biodiversity, areas with loss of biological corridors as well as socio-economic information, permitting contextualisation of the problem of mining on

the affected territories, mainly those considered vulnerable (Li et al., 2000; Lara-Rodríguez et al., 2020, Cheng et al., 2020). However, the use of satellites with better spectral and spatial resolution is required to improve the said classification in order to specify, from a retrospective and prospective scale, the impact of mining generated on these areas (Ozelkan et al., 2012). Approaches such as those described have been proposed by Efe et al. (2012).

In this same sense, Lysaniuk et al. (2021) were able to estimate the spatial and occupational risks that chronic exposure to pollutants would generate from a geological and spatial perspective on the vulnerable population at risk of exposure. The above is possible by considering GIS as a feasible tool to improve surveillance and control systems (Zhou, 2021), especially in the areas where there is an urgent need to use continuous monitoring plans (UNODC, 2020).

Geographic information systems have proven effective for the identification of mining contaminated sites in the Turkish region through spatial terrain classification (Yenilmez et al., 2011). The results that not only managed to reduce the number of samplings (with the consequent decrease in costs when defining the level and degree of contamination of soils and affected water bodies) but also allowed the data to be taken into account for the development of prospective studies. The above results were based on topographic distribution, runoff routes, and in situ determination of contaminant levels in the area.

Based on the previous analysis, it was possible to find an inversely proportional relationship between the sites closest to the point sources and the equidistant water bodies (Yenilmez et al., 2011). The data and research that would be relevant for the evaluation of pollutants from open-pit mining in Colombia due to the large number of pollutants resulting from open-pit mining activities that originate in the country (Lara-Rodríguez et al., 2020), for example, the emission of pollutants, such as mercury, in some regions of Colombia (UNODC, 2020).

Yucel et al. (2014), on the other hand, managed to evaluate the edge effect that mining would have on using satellite images in time series on the areas having the presence of acid lakes generated by mining. The results of this study showed that the presence of these areas caused a series of related environmental risks due to the harmful physicochemical properties that they would have on the surrounding soils and watersheds, generating in parallel an increase in the loss of land area of these sites.

Identifying the sites where legal and illegal activities are carried out has a strong impact on how this problem is addressed. For this reason, obtaining environmental databases turned relevant to assess the state of contamination of territory with presence of mining. For example, in Morocco, Khalil et al. (2014) managed to map nearly 200 mining sites for the space-time evaluation of the impact of abandoned mines in this territory, finding that stream networks and pollutant dispersion mechanisms would vary widely depending on the database used. Hence, it is proposed that this GIS be updated frequently in order to generate more reliable information for the development of more robust spatial analyses (Grenon and Hadjigeorgiou, 2010).

7 Future Directions of GIS in Open-pit Mining

The relationship between the applicability of the use of information technologies with the clean, safe and efficient production of anthropogenic activities and the environment is becoming more and more evident. In this sense, the use of unmanned technologies has allowed in places, such as Fankou in China, the exploitation of important minerals with the technology based on wireless communication, thus improving competitiveness and consequently reducing the adverse effects that the same activities would have under the development of other conventional ways of production (Li and Zhang, 2018). In this sense, countries such as Finland, Sweden and Canada have opted for this intelligent and fully automated in situ production initiative, reducing the amount of waste and improving the use and optimisation of resources (Walker, 2012). Technology that would be worth opting for in countries such as Colombia, where extraction systems that are increasingly friendly and sustainable with the environment are required.

Other studies based on advances in technological development have allowed the recovery of open-pit mining areas for arable land using GIS and multi-criteria evaluation methods to determine their suitability for restoration processes (Baoying, 2012). In the case of Colombia, automated models based on remote sensing have been developed in recent years using learning algorithms that are able to predict mining areas linked to open pits in the national territory. In this case, Saavedra (2022) achieved the implementation of high spatial resolution satellite data (such as the 5-m NICFI PLANET satellite) and the use of radar satellites (Sentinel 1) to generate the development of high prediction models, through training frameworks, with an accuracy greater than 80% on Colombian territories. The results of this study allowed this methodology to be suggested as the first step for advancing the development of strategies based on artificial intelligence in the territories impacted by illegal mining at the national level (Saavedra, 2022). In the same sense, it is proposed that continuous improvements in GIS and the adequate use of remote sensors would improve the analysis of environmental data with spatial information through remote sensing (Fengxia and Wenbao, 2010).

From a global perspective, it is necessary to improve the access and updating of spatial databases to optimise the thickness of the information obtained (Li et al., 2000). Likewise, it is evident to promote the use of geographic information technologies to improve access and availability of data in real-time that could be used as input for the development of sustainable mining, mainly in the regions of the territory that have been historically impacted or where climatic conditions make it difficult to identify them directly.

Likewise, it is proposed that the use of automated learning models, risk assessments, and monitoring of both legal and illegal mining concessions would be the basic inputs to achieve the sustainable development objectives (12 and 15) that point out to perpetuate the care of the environment by minimising the impacts that mining would generate on these territories (Argota, 2017). These are the objectives for which Colombia must make efforts to obtain action plans to mitigate its impact (UNODC,

2020). It is necessary to know the impacts generated by mining directly or indirectly, for which any data reported about this activity must have GIS-based spatial support that manages to dimension and correlate the data obtained, and thus be taken into account for future research (Castellanos-Quiroz et al., 2017).

In the case of Colombia, in particular, open-pit mining is concentrated mainly in territories such as Antioquia, Chocó and Bolívar, which correspond to 87% of the national total available mining regions. This spatial distribution is more evident in municipalities such as Zaragoza, Nechí and Novita, which represent 22% of the mining territory in alluvial deposits (UNODC, 2020). Hence, conducting remote sensing research for mining in Colombia should take precedence over these zones. Unfortunately, and under a discouraging scenario, Colombia intends to double gold production by 2030 because of the profitability that this activity generates in its economy. However, it is proposed that the stage presence and the formalisation of this activity in many territories should increase in parallel with its operations in order to mitigate the socio-environmental problems that are usually related to mining (Lara et al., 2020).

Under this scenario, it is necessary to increase development in the advancement of perception tools, the number of investigations as well as in advancing the use of both open source and paid GIS software. This provides more information that can be taken into account for the implementation of action plans against mining activities that have not been formalised, taking into account the geographical aspects and the socio-economic conditions of each territory (Alegria et al., 2011). The foregoing is mainly in the areas where socio-economic and environmental dynamics are usually complex (Lara-Rodríguez et al., 2020) in the short, medium and long terms (Blom et al., 2019).

For some years, different planning software have been developed to face the challenges in which miners are involved in the execution of their activities, to minimise their impact, thus achieving an easy exchange of data between GIS and mine-planning software (Elroi and Price, 1998). Similarly, different software modules have been proposed as an alternative to the commercial packages currently used in industrial mining ArcGIS Pro, Vulcan or Mining Visualization System (MVS), managing to become accessible to small mining operators by offering them information regarding the risks of their activities. For example, exposure to noise, and the direct or indirect impacts that their activities would generate on surface and groundwater, among others, thus achieving greater production control through real-time GIS integration of their production processes (Zarubin et al., 2021). These methodologies, although not of recent appearance, would be feasible for implementation in some Colombian territories, mainly in vulnerable territories (UNODC, 2020).

Finally, it should be noted that the use of remote sensing images has been expended currently to determine the space-time variations of mining processes that support decision-making and territorial planning, thus improving their transparency (Choi et al., 2020, Risk et al., 2020). Thus, these would manage to have a beneficial impact on ecosystem services and biodiversity loss, components in which mining has managed to demonstrate a direct impact (Risk et al., 2020) and in which the GIS seems to have an important potential for the regulation and control of the same.

8 Conclusions

Geographic information systems are currently a key tool for understanding the dynamics and processes that take place in mining environments, as well as their advantages and disadvantages, the latter being the most documented and unfavourably linked factors to the development of the territories, mainly when it is interrelated with illegal processes. Based on the existing information on the use of GIS in mining at a global level, it can be concluded that its use, together with that of remote sensors, widely enables the processes of control, monitoring and surveillance of activities related to mining, thus facilitating the decision-making. It is important for Colombia to make first approximations on mining satellite data and GIS that are currently available, but not before making a call to better document the processes that occur in the territory, including those of difficult georeferencing, such as the Colombian Pacific areas. This is in order to take measures to mitigate their impact through their evaluation and monitoring, as has been developed in the world and as has been briefly described in this chapter.

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Chapter 3

Pollution Assessment of Soil and Groundwater Resources Around the Gold and Chromite Mining Regions of Southern India



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and Manavalan Satyanarayanan

1 Introduction

The environment combines all external factors that affect an individual's life or a population of living organisms (Liu et al., 2021). Man is exploiting the environment by destroying natural resources. Natural ecosystems were balanced and remained unaffected since the dawn of civilization. The uncontrolled human population has disturbed everything and created a negative impact on the environment through environmental pollution, causing a diagnostic change in the characteristics of air, land and water, which affect all living species on the planet (Sherbinin et al., 2007).

The problems of environmental degradation have evoked global concern in recent years. These problems have mainly arisen due to resource exploitation, population growth and poverty, industrialisation, urbanisation, and a spurt in energy consumption. Among the key economic activities, mining has come under severe public criticism because of accompanying eco-degradation. It is known to cause forest degradation, soil erosion, choking streams and rivers, and air and water pollution. It is thus responsible for ecological imbalance, which has made the ecosystem unsustainable (Chaudhary, 1992; Gautam and Patra, 2015).

Mining methods adopted for extracting the ore body stipulate the degradation rate of rock and soil (Trivedy and Sinha, 1990). Opencast mining methods are adopted for shallow ore bodies, preferably before groundwater level, while underground methods are used for concealed ore bodies, especially for wining precious metals (Brown, 1990). Various studies have suggested that quality of water is affected in both open

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cast and underground mining areas because of mechanized mining methods, ground alternation, water flow and air quality (Ripley et al., 1996).

2 Impact of Mining on the Environment

Deterioration of the environment in terms of air, soil and water pollution is wittingly or unwittingly due to the contributions of the mining industry. The main environmental problems associated with mining activities are as follows: (i) water pollution because of removing of overburdened wastes, discharge of mine waters, discharge of tailings into water bodies, mining drainage, and coal washing operations; (ii) air pollution because of release of harmful gases into the atmosphere and generation of dust during various operations; (iii) accelerated land degradation because of large-scale excavation, disposal of mine waste and tailings and subsidence; (iv) structural alterations in elevated terrain, particularly during open cast mine operations causing landslides; (v) deforestation due to felling of trees for the construction of roads and fuel wood; and (vi) disruption of water regime caused by trenching and stripping of land.

India is endowed with a wide range of mineral reserves, and the total area held under mining leases of the working mines is 7,453 km², and most of the minerals originate from small-scale mines (Indian Bureau of Mines [IBM], 2018). While a substantial part of this area comprises underground mines, the existing opencast excavations cover an average of 5% of the area (Chaudhary, 1992). According to Valdiya (1987), about 80,000 ha of land is not only under the stress of mining activities of various types but also disturbed by other mining-associated activities.

3 Heavy Metal Pollution

Indiscriminating mining of natural resources and waste disposal are this country's primary sources of heavy metal pollution. Weathering and subsequent heavy metal leaching are potential environmental risks (Riccardo et al., 2009). Metal contamination of soil and groundwater because of mining and associated activities at zinc and chromite mines has been reported in India (Prustry et al., 1994; Krishna et al., 2013; Dhakate and Singh, 2008). On the other hand, excavated and abandoned mines, left without reclamation, also endanger the environment. The magnitude of the impacts from past mining is often considered, because it represents thousands of mining sites that continue to pose a potential threat to human safety and health and/or environmental damage. Metals associated with gold (Au) and silver (Ag) mines, including cadmium (Cd), copper (Cu), lead (Pb), arsenic (As) and zinc (Zn), are dispersed downstream due to the weathering process of tailings. Thus, the extent and degree of trace metal contamination around the mines vary, depending upon chemical characteristics and mineralization of tailings. The impact of mining on soil

and groundwater regimes can be evaluated using hydrogeological investigations and groundwater quality assessment.

4 Case Studies

4.1 *Assessment of Soil Contamination at Kolar Gold Mining Site*

4.1.1 Study Area

Kolar is located south of Bangarpet taluk in Karnataka, India, at latitude 12°57' N and longitude 76°16' E. The sample location map is shown in Fig. 1. The Kolar schist belt of south India is renowned for its gold deposits and has been mined consistently for more than 100 years. Kolar Gold Field (KGF) mines are located in the semi-arid region of the Southern plains of Karnataka. The area is characterized by an annual rainfall of 740 mm. The warm day temperature spread over most parts of the year is around 27–35°C (Rao and Venkatramana Reddy, 2006). The gold lodes of KGF lie in a narrow band of rocks of the Dharwar series. The Kolar schist belt is a north–south trending ~80 × 2–4-km volcanic-dominated Archean schist belt in the eastern block of the Dharwar craton. The belt predominantly consists of ~2,700-Ma mafic metavolcanics represented by four texturally distinct amphibolites.

In the Kolar schist belt, gold occurs in gold quartz sulfide lodes and gold quartz calcite vein systems. Gold sulfide lodes include significant quantities of magnetite, ilmenite and graphite. A well-foliated felsic unit, known as the 'Champion Gneiss', lies along the eastern part of the belt and includes xenoliths of granitoid, amphibolites, and banded iron formation at places as well as vein quartz. The host rock of gold is the hornblende schist of the lower Dharwar age, accompanied by pegmatites and Champion gneisses. The three main mines of KGF are Nundidurg, Champion Reef and Mysore Mines (Arora et al., 2001).

4.1.2 Sample Collection, Preparation and Analysis

A total of 60 soil samples were collected in May 2006. The area covered includes the old mining site and the surrounding agricultural and residential soils. After removing surface contamination, soil samples were collected from 5–15-cm depth of the outer surface. The sampling was conducted using a plastic spatula and no metal tools were used. Polythene bags with self-locking seals were used to collect the samples, and they were sealed twice. Samples were dried for two days at 60°C. The dry soil samples were crunched into a fine powder (US Standard) in a swing grinding mill. X-ray fluorescence (XRF) spectrometry was used to analyse trace and major elements in sample pellets prepared in collapsible aluminum cups with a boric acid backing.

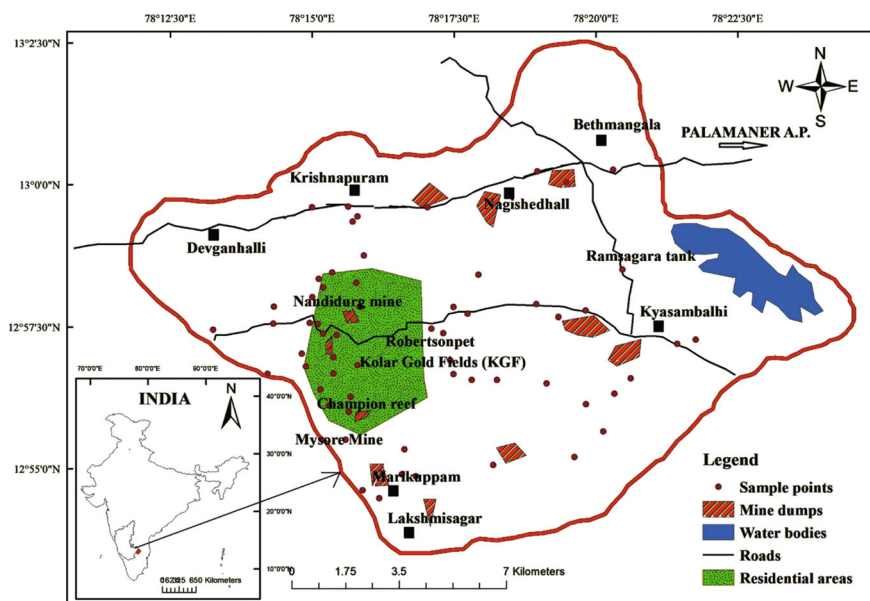


Fig. 1 Sample location map of the study area – Kolar Gold Field (KGF), Karnataka, India.

The pellets were prepared for XRF analysis using hydraulic press (Krishna et al., 2007).

In order to determine major oxides and trace elements in soil samples, an X-ray Fluorescence Spectrometer, Philips MagiX PRO Model PW 2440, coupled with an automatic sample changer, was used. A goniometer-based measuring channel covered the entire elemental range on MagiX PRO. A microprocessor controlled the instrument to ensure maximum flexibility. This experiment used a rhodium (Rh) anode X-ray tube, which could operate at 60 kV and 125 mA at a maximum power of 4 kW. XRF was calibrated using international soil reference material, such as SO-1, SO-2, SO-3 and SO-4, procured from Canadian-certified reference material projects, and SRMs 2709, 2710 and 2711 of the National Institute for Standards and Technology, MD, USA.

4.1.3 Results and Discussion

The concentrations of heavy metals in the soils of KGF, and their descriptive statistics along with threshold limits (Canadian Council of Ministers of the Environment [CCME], 1999), are presented in Table 1. Arsenic is found in the range of 0–83.7 mg/kg, although the average is 47 mg/kg. The maximum threshold limit recommended in soils is 12 mg/kg. Figure 2a shows the distribution map of arsenic. High values were observed in the samples taken from soils nearer to the dumps and water bodies near the mining area. Arsenic is high due to the presence of arsenopyrite

Table 1 Statistical summary of heavy metal concentrations (mg/kg) in the soil samples of Kolar Gold Field (KGF) mining area, Karnataka, India ($n = 30$)

	<i>As</i>	<i>Ba</i>	<i>Co</i>	<i>Cr</i>	<i>Cu</i>	<i>Ni</i>	<i>Pb</i>	<i>Sr</i>	<i>V</i>	<i>Zn</i>
Maximum	83.7	998.6	96.2	500.4	128.9	202.4	198.0	374.9	295.4	202.3
Minimum	0.0	462.7	0.5	39.5	11.9	4.6	17.2	80.7	17.1	28.1
Average	47.0	720.8	41.0	184.9	59.2	77.0	125.1	144.2	144.0	69.0
Median	11.3	722.2	50.1	178.6	69.6	89.7	53.2	120.2	157.1	66.7
Standard deviation	128.9	131.7	31.6	156.4	32.9	60.9	266.6	60.8	89.9	30.3
Threshold value	12.0	500.0	50.0	64.0	63.0	50.0	140.0	200.0	130.0	200.0

present in the mining area (Siddaiah and Rajamani, 1989). Such abnormal values of arsenic could be attributed to the leaching of the dumps, as the concentrations were high in the samples collected among the dumps. The mine dumps characteristically contain arsenic, which is mainly present as arsenopyrites in the gold ore. The samples collected from the nearby agricultural areas were found to be low in arsenic. This contamination could be due to the presence of barite minerals, typically found with gold deposits because of their significant association with chalcopyrite. Cobalt ranged from 0 to 96.2 mg/kg with an average of 41 mg/kg. Except few, all soil samples collected from the nearby residential area were found to have cobalt higher than the background concentration limit of 50 mg/kg. The distribution of cobalt is shown in Figure 2b. Cobalt is an important component of pyrite and pyrrhotite. The contamination was predominantly due to the leaching of dumps, as we discovered high concentrations nearby. The concentration of cobalt is very low in agricultural lands, so the average is less.

Chromium has a minimum concentration of 39.5 mg/kg and a maximum of 500.4 mg/kg, with an average concentration of 184.9 mg/kg. The background concentration limit of chromium in soil prescribed by the Canadian Soil Quality Guidelines is 64 mg/kg. The soils of KGF were found to have a higher average concentration than the threshold limit. The contamination of chromium followed the pattern of other elements, with higher values being concentrated near the mining area. Figure 2(a)–(d) shows the distribution of arsenic, cobalt, chromium and lead in the soils. Chromium was proven as high in almost all the ore deposits of KGF, which could be associated with its elevated values in the soils of KGF.

Copper had an average concentration of 59.2 mg/kg, varying from 11.9 to 128.9 mg/kg. The average, found to be higher than its background concentration limit in soils, was 63 mg/kg. None of the samples near the mining area had copper concentrations less than 60 mg/kg. Copper contamination could be due to the presence of mine tailings rich in chalcopyrite, spharelite and galena.

Lead concentration was more than the threshold value in the samples collected from near the mining site. The background soil concentration of lead according to the Canadian Soil Quality Guidelines is 98 mg/kg. Lead concentration was found in

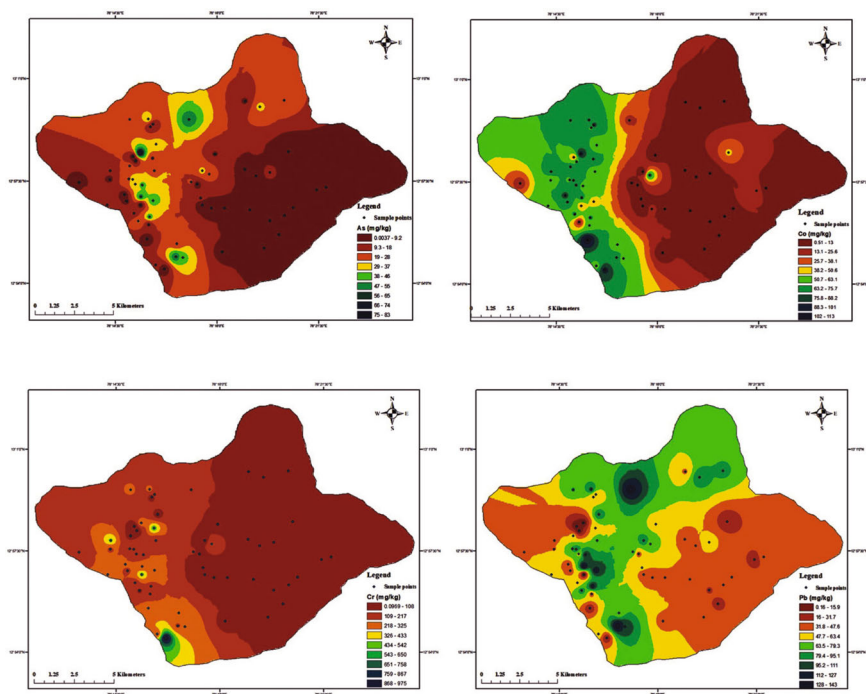


Fig. 2 Spatial distribution of (a) arsenic, (b) cobalt, (c) chromium and (d) lead in the soils of Kolar Gold Fields (KGF) in Karnataka, India.

the range of 17–198 mg/kg, the average being 125 mg/kg (Table 2). Figure 2d shows the distribution of lead. Chalcopyrite, sphalerite and galena are the main sources of lead in the ore deposits of KGF. Away from the mining site, lead concentrations were found lower than the threshold limit. It exceeded the limit of 1,000 mg/kg in two soil samples that were very close to each other as well as close to the dump site.

4.2 Evaluation of Groundwater Quality at the Nuggihalli Chromite Mining Area

4.2.1 Study Area

The Nuggihalli schist belt and its surroundings are located in the Hassan province of Karnataka, India. The study area is geographically located at latitude 12°54′58″–13°1′48″ N and longitude 76°29′50″–76°32′24″ E and covers an area of 400 km² (Fig. 3). The prominent chromite mineralized sites in the study region are Jambur, Tagdur and Byrapur. Also, agricultural and residential zones were demarked to assess the groundwater quality. Geologically, chromite ores occur in these areas as lensoid

Table 2 Statistical summary of the measured water parameters of the Nuggihalli mining area in Karnataka, India, with comparison to WHO (1993) standards for drinking water

<i>Water quality parameter*</i>	<i>Most desirable limits</i>	<i>Max allowable limits</i>	<i>No. of samples exceeding allowable limits</i>	<i>Proportion of samples exceeding allowable limits (%)</i>
pH	6.5–8.5	9.2	Nil	Nil
Electrical conductivity (EC)	750	1,500	6	18.18
Total dissolved solids (TDS)	500	1,500	3	9.09
Ca ⁺⁺	75	200	1	3.03
Mg ⁺⁺	50	150	1	3.03
K ⁺	0	12	2	6.06
Na ⁺	0	200	Nil	Nil
Cl ⁻	200	600	Nil	Nil
HCO ₃ ⁻	0	300	28	84.84
NO ₃ ⁻	0	50	6	18.18
SO ₄ ²⁻	45	400	Nil	Nil
F ⁻	0	1.5	4	12.12
TH	100	500	8	24.24
As	0.01	0.05	23	69.6
Fe	0.3	No relax.	Nil	Nil
Mn	0.1	0.3	8	24.24
B	0.50	1.00	Nil	Nil

* All parameter in mg/L; EC in uS/cm, except pH.

bodies and pods, which are discontinuous, but intermittently occur all along the schist belt. In general, occurrence, movement and storage of groundwater are influenced by lithology, thickness and structure of rock formations. Weathered and fractured granites, granitic gneiss and chromite-bearing ultramafic rocks are dominant aquifers in the Nuggihalli schist belt. Although there is no river in the study area, it is very well controlled by drainage and surface water bodies.

4.2.2 Sample Collection, Preparation and Analysis

In May 2011, a total of 33 groundwater samples were collected in separate 1-L and 250-mL polythene bottles from different wells of the study area. The 250-mL bottle was acidified with HNO₃. pH value, electrical conductivity (EC) and TDS of water samples were measured *in situ* using portable meters (Eutech pHTester10,

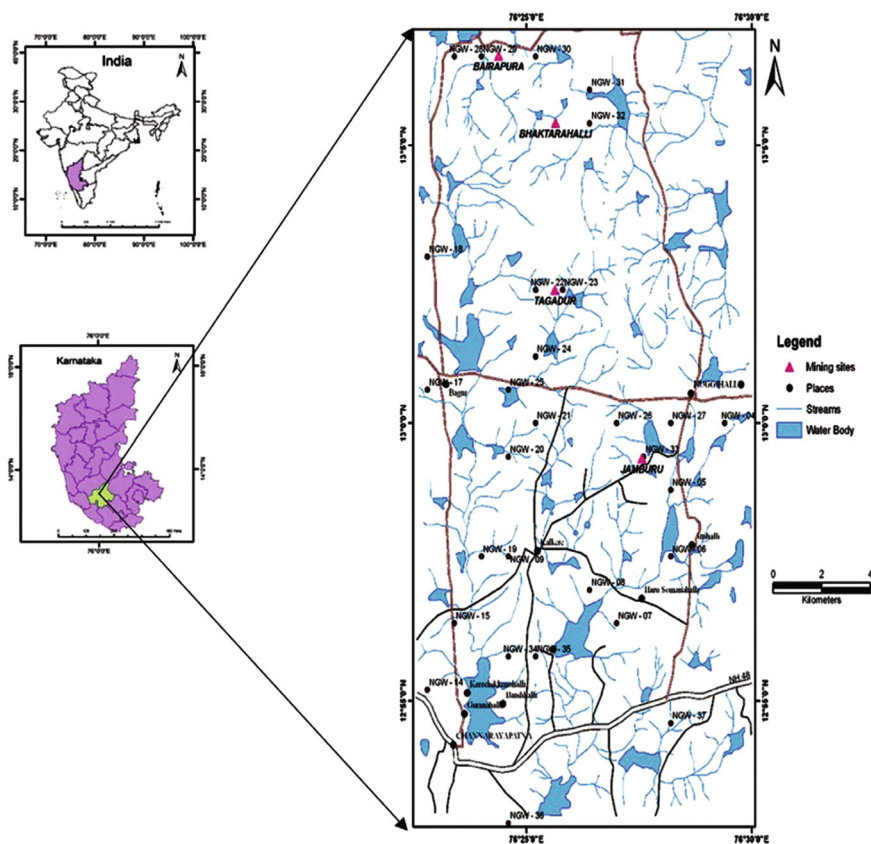


Fig. 3 Map showing the Nuggihalli chromite mining area of Karnataka state, India, including sampling locations of groundwater.

ECTester11⁺ and TDSTester11⁺). Major cations, such as calcium, magnesium, sodium and potassium, (ASTM, 2007) and anions, such as chloride, nitrate, sulphate and fluoride (United States Environmental Protection Agency [USEPA], 1999), were analyzed by ion chromatography (Metrohm, 882 Compact IC plus). The acidified water samples were analysed for the concentration of heavy metals using an inductively coupled plasma optical emission spectrophotometer (Perkin Elmer, ICP-OES, Optima 4300 DV). In general, groundwater quality is affected by age of pumps and natural discharge. Further, the water was heterogeneous due to recharge from precipitation and contact with different types of rocks. Hence, a good number of samples were collected over the entire area to determine the study area's water quality.

4.2.3 Results and Discussion

Hydrogeochemistry of groundwater: Water–rock interactions and anthropogenic activities primarily control the groundwater hydrogeochemistry; therefore, understanding the hydrogeochemistry of groundwater is vital. Table 2 shows summary of the data of various parameters, such as pH, EC, TDS, and major anions and cations of groundwater. The pH value was observed with an average of 7.3, indicating that groundwater in the study area was alkaline. The average EC value was 1,103 $\mu\text{S}/\text{cm}$, while the average TDS value was 900 mg/L during pre-monsoon season. The data concerning alkali and alkaline earth reveals a high concentration of Ca^{2+} , with the following order $\text{Ca} > \text{Mg} > \text{Na} > \text{K}$. Among the various anions determined in groundwater samples, HCO_3^- , Cl^- and SO_4^{2-} were the major anions identified in the following order: $\text{HCO}_3^- > \text{Cl}^- > \text{SO}_4^{2-}$. Dissolution of carbonates in the rocks could have appended to the groundwater system because of irrigation and rainfall infiltration., (Kurakalva et al., 2021). Chemical data of the groundwater samples was plotted on a Piper (1994) trilinear diagram in order to understand geochemical properties. The ionic composition of groundwater was plotted in two trilinear and one diamond-shaped field of the Piper diagram. Consequently, water was divided into six hydrochemical facies, and their corresponding graphs are shown in Fig. 4. The plot shows that groundwater is a predominant CaHCO_3 type of water. The study area was significantly dominated by the CaHCO_3 type of water. This could be due to the domination of mineral constituents and rock weathering in addition to recharge activities and induced anthropogenic activities that influenced the groundwater chemistry.

Hydrogeochemical evolution: The chemical composition of groundwater could be interpreted through rock–water interactions and subsequent chemical reactions. In addition, aquifer materials also play a major role in the evolution of water chemistry. Thus, the analytical data presented in the Gibbs (1970) plots demonstrated water–rock interactions. The data for both seasons was plotted between Gibbs ratios for cations $\text{Na}+\text{K}/\text{Na}+\text{K}+\text{Ca}$ and their respective TDS values of groundwater samples. The plot (Fig. 5) revealed that around 65% of the groundwater samples belonged to the dominant rock category, and the rest diminished in evaporation. It explained that most groundwater samples happened under a dominant rock field caused by the interaction between aquifer rocks and groundwater.

Water quality assessment: Table 2 compares the chemical analysis of groundwater data to the standard guideline values recommended by the World Health Organization (WHO, 1993) for drinking purposes. It included the most desirable limits and maximum allowable limits of various parameters. According to the Bureau of Indian Standards (BIS, 1991), the desirable limit of pH for drinking purposes is 6.5–8.5. In the study area, most of the groundwater samples had a pH value within the prescribed limits, indicating its suitability for drinking. Classification of groundwater EC of the study area is given in Table 3 and its spatial distribution is shown in Fig. 6(a). It was found that 81% of the groundwater samples were within the permissible limits; however, 18% of the samples were not within the permissible limit and were marginally poor in quality according to the WHO (1993) standards.

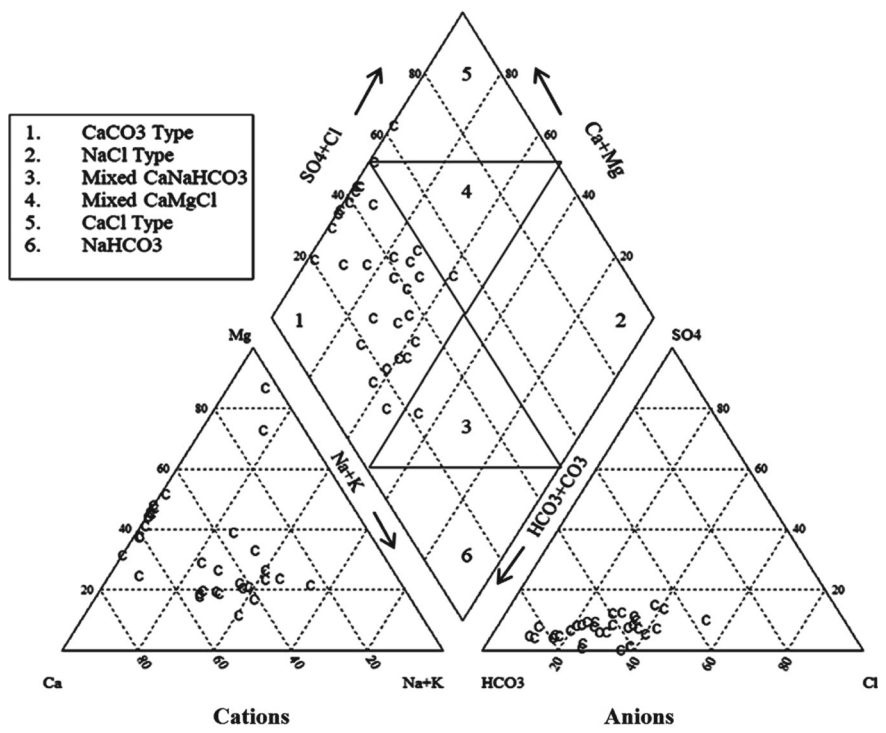


Fig. 4 Piper diagram depicting hydrochemical facies of groundwater of the Nuggihalli mining area in Karnataka, India.

Fig. 5 Gibbs diagram for groundwater quality.

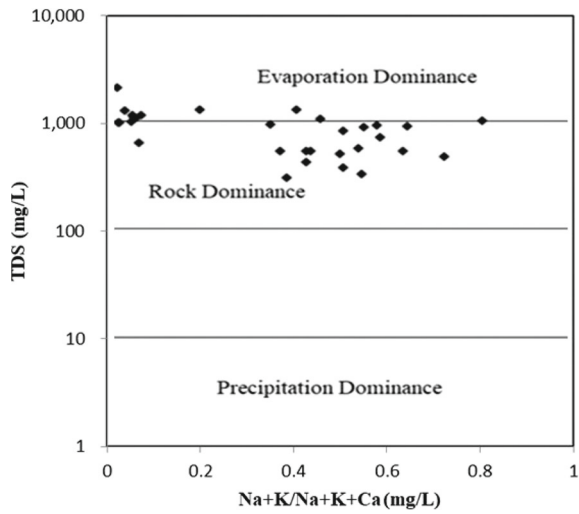


Table 3 Classification of groundwater based on electrical conductivity (EC), total dissolved solids (TDS) and total hardness values at the Nuggihalli mining area in Karnataka, India

<i>Parameters</i>	<i>Range</i>	<i>Classification</i>	<i>No. of samples exceeding allowable limits</i>	<i>Proportion of samples exceeding allowable limits (%)</i>
EC ($\mu\text{S}/\text{cm}$) (WHO, 1993)	<1,500	Permissible	27	81.81
	250-750	Not permissible	6	18.18
	750-2,000	Hazardous	Nil	Nil
TDS (mg/L) (Davis and Dewiest, 1966)	<500	Desirable for drinking	6	18.18
	500-1,000	Permissible for drinking	15	45.45
	1,000-3,000	Useful for irrigation	12	36.36
	>3,000	Unfit for drinking and irrigation	Nil	Nil
TDS (mg/L) (Davis and Dewiest, 1966)	<1,000	Fresh water type	21	63.63
	1,000–10,000	Brackish water type	12	36.36
	10,000-100,000	Saline water type	Nil	Nil
	>1,00,000	Brine water type	Nil	Nil
Total hardness (Sawyer and McCarthy, 1967)	<75	Soft	Nil	Nil
	75–150	Moderately high	4	12.12
	150–300	Hard	11	33.33
	>300	Very hard	18	54.54

The contour map of TDS in groundwater is shown in Fig. 6(b). It demonstrated that only 18% of the samples had TDS below 500 mg/L (Davis and DeWiest, 1966) and could be used for drinking purposes without any risk (Table 3). Further, the results demonstrated that following the Freeze and Cherry's (1979) classification, 63% groundwater of the area was of freshwater type, and 36% of the samples represented brackish water (Table 3).

The total hardness of groundwater samples in the present study exhibited minimum content of 109.74 mg/L and a maximum of 837.11 mg/L, with an average of 369.15 mg/L. The spatial variation of hardness in groundwater samples (Fig. 6c) demonstrated higher concentrations in the eastern part of the study area (i.e. Jambur abandoned mining area and the surroundings). According to the Sawyer and Macartly (1967) classification, 33% of groundwater samples were in the hard category; similarly, 54% of the samples were in the very hard category (Table 3) and were not suitable for drinking purposes. The analytical data on cations demonstrated that concentration of 6% of the samples with K^+ and 3% of the samples with

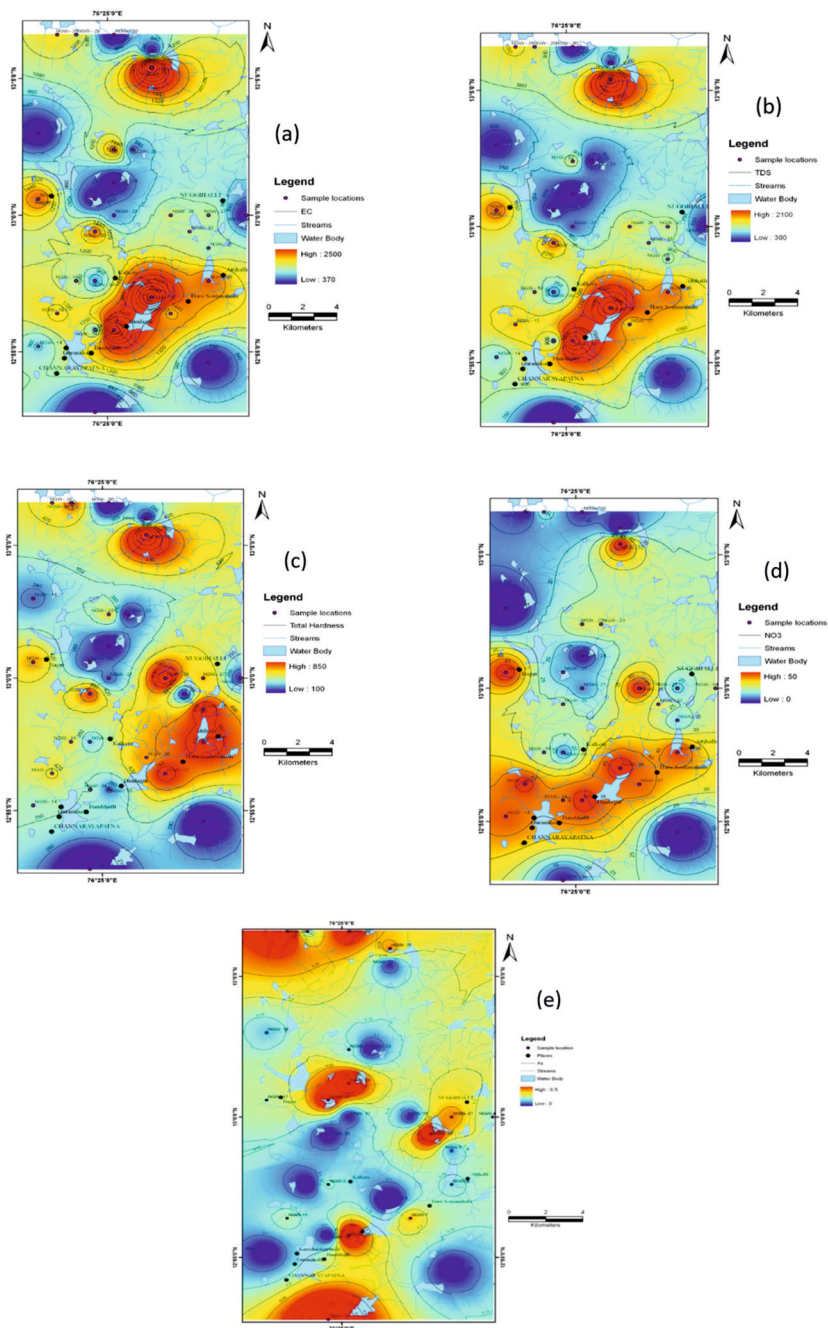


Fig. 6 Spatial distribution of (a) electrical conductivity (EC), (b) total dissolved solids (TDS), (c) total hardness, (d) nitrate and (e) arsenic in groundwater of the Nuggihalli chromite mining area in Karnataka, India.

Ca^{2+} and Mg^{2+} exceeded the WHO (1993) limit, but Na^{+} concentration was within the permissible limits. This could be due to the ion exchange process occurring in aquifer.

Spatial variation in the nitrate concentration of groundwater of the study area is shown in Fig. 6d. Approximately 24% of the total groundwater samples collected exceed the WHO permissible limit of 50 mg/L nitrate concentration. It varied from 0.30 to 49.90 mg/L, with an average of 26.18 mg/L because of excessive fertilizer use on agricultural land (Jalali, 2005).

Fluoride concentration varied between 0.80 mg/L and 2.00 mg/L, with an average of 1.18 mg/L. In all, 12% of the total groundwater samples collected in the study region exceeded the WHO permissible limit of 1.5 mg/L of fluorides.

Chloride concentration in the groundwater samples varied between 22.70 mg/L and 524.70 mg/L, with an average of 138.62 mg/L observed within a permissible limit. Sulphate concentration in the study area was within the permissible limit of 400 mg/L as per the WHO standards. It varied between 1.90 mg/L and 141.90 mg/L, with an average of 47.27 mg/L. High sulphate concentration in drinking water is toxic and causes laxative efficacy.

Heavy metal contamination: A significant concentration of arsenic in groundwater was observed among various heavy metals. The present study demonstrated that almost 79% of the examined wells exhibited arsenic concentrations higher than the WHO permissible limit of 0.010 mg/L for drinking purposes (Table 3). The concentration of arsenic ranged from 0.002 to 0.51 mg/L. The spatial distribution of arsenic in the groundwater of Nuggihalli and its surrounding areas, southern Karnataka, India, is illustrated in Fig. 6e. The highest concentrations of arsenic were discovered in well numbers 24, 33, 35 and 36, with concentration ranging from 0.43 to 0.51 mg/L. These wells also had high concentrations of potassium, sodium, TDS, bicarbonates, chlorides, iron and boron, but a minimum concentration of nitrates.

Correlation coefficients were calculated to examine possible relationships among the measured parameters of groundwater samples. The plots revealed that arsenic exhibited a significant correlation ($R^2 = 0.84$) with iron and manganese, indicating a common geogenic origin of these elements (Fig. 7). It is evident from literature (Ayotte et al., 2003) that the correlation between arsenic and iron/manganese is frequently observed in groundwater because, under certain conditions, the presence of iron/manganese oxyhydroxides could lead to desorption of arsenic, iron and manganese. However, it was observed that wells with high arsenic concentration also exhibited high concentration of boron, suggesting a common origin of both elements, which is also reported by Bhattacharya et al. (2006). Boron dissolution in groundwater is generally geogenic and related to hydrothermal or saline water resources, water-rock interaction processes, and sea water intrusion (Pennisi et al., 2006). However, the concentration of boron in groundwater samples was within the WHO (1993) permissible limits for drinking.

The results suggested that these relations of common geogenic origin and condition could enhance the mobility of arsenic in the groundwater. Further, the groundwater samples collected from Chikkondanahalli and Bairapura wells exhibited the

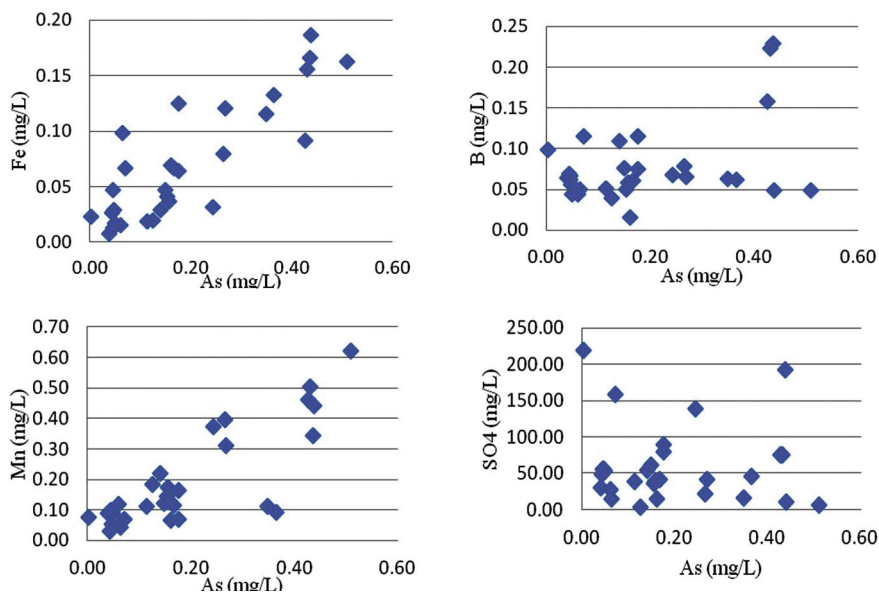


Fig. 7 Scatter plot showing the correlation between As and Fe, Mn, B and SO_4 .

highest arsenic concentrations because both locations were near the chromite mining areas.

5 Conclusion

The study conducted at the Kolar gold mining area demonstrated that soils in the vicinity of mining dumps were significantly contaminated with metals, such as arsenic, cobalt, chromium, copper, vanadium, nickel and lead, far above the threshold permissible limits. Because of the residential area amidst mining dumps, a high probability prevails that these high concentrations could cause health hazards. Barium had high concentrations at almost all sampling sites, irrespective of the location. It did not indicate any correlation with mining locations. Hence, the source of contamination could be limited as geogenic.

The average of some elements, such as strontium and zinc, was found less than the threshold limit but others, such as arsenic, cobalt, chromium, copper, vanadium, nickel and lead, were found in high concentrations near the mining site. The sources of contamination of these elements are dumps through anthropogenic activity. These dumps are exposed to the atmosphere without proper treatment or capping. Therefore, trace elements are leached from dumps to the soil, always leading to a possible contamination of water resources. Remediation of these tailings for the removal of

high concentrations of trace elements must be done, or at least capping must be done with trees to stop their weathering.

Groundwater quality at the Nuggihalli chromite mining site identified six major hydrochemical facies, thus elucidating CaHCO_3 type of water. Arsenic showed a significant correlation with iron and manganese, indicating a common geogenic origin of these elements. Stringent monitoring and control measures in the regions of low groundwater quality are necessary to ensure sustainable and safe use of the resource.

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Chapter 4

Impact of Heavy Metal Pollution on Soil and Groundwater Quality in the Vicinity of Coal Mines



Deep Raj and Kousik Das

1 Introduction

In the past 20–30 years, deterioration in soil quality has become a serious environmental issue. Rapid industrialization and mining activities have triggered the deterioration of soil quality. Owing to these anthropogenic activities, soil is exposed to contaminants, such as heavy metals (HMs) (Xiao et al., 2020; Raj and Maiti, 2019), which, in turn, result in the degradation of soil quality as well as soil fertility. HMs such as mercury (Hg), arsenic (As), cadmium (Cd), chromium (Cr) and lead (Pb) because of easy accumulation and non-degradable properties are highly toxic, and their impact on soils is hazardous (Wang et al., 2020). The presence of HMs also damages the useful microbes of soil. Because soil acts as a medium for plant growth, HMs are also accumulated in plant cells through the adsorption of metals, which damage the cellular components of the plant system (Chen et al., 2019). Hence, the soils containing HMs affect the surrounding ecosystem of coal mines.

Over the past few years, numerous studies have been conducted on soil pollution caused by the release of HMs because of coal mining activities. For example, Marove et al. (2022) assessed HM concentration in the soil samples of active coal mine in Moatize district, Tete Province, Mozambique. The authors observed that hazardous metals from soils around coal mines were getting leached in acidic environment, and metals, such as manganese (Mn), are also bioavailable to plants. Further, the study concluded that the bioavailability of metals could present serious health risks to the human population as well as the entire ecosystem. In another study, Ge et al. (2016) assessed ecological risk (ER) from different HMs (Cd, Hg, As, copper [Cu], Pb,

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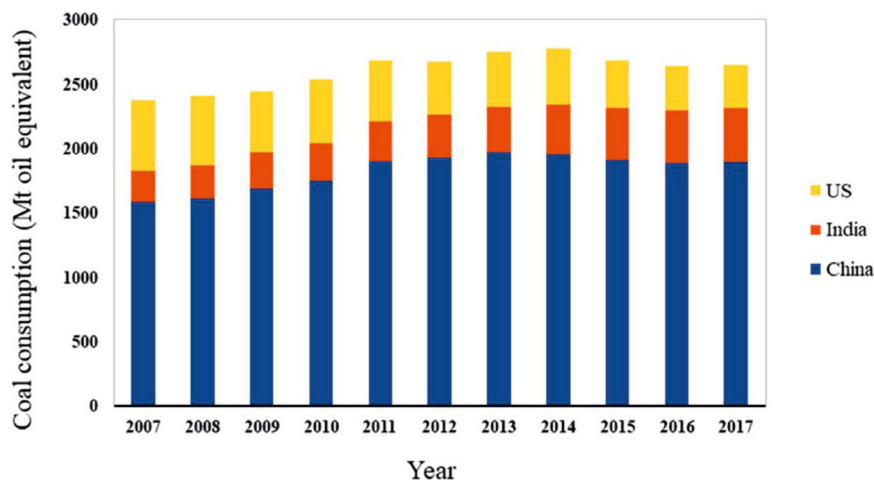


Fig. 1. Coal consumption in China, India, and the United States (2007–2017). (Source: Dudley, 2018)

Cr, and zinc [Zn]) that were available in the surrounding soils of a coal waste pile at Naluo Coal Mine, Guizhou, China. The results of the cited study demonstrated that the strongly elevated concentrations of HMs caused higher ecological risk in the affected area. In India, the coal mining industry plays an important role in its economy, that is more than 70% of total power generated is from coal. The contribution of coal is more than half in the scenario of total energy requirement, and coal demand is growing much faster than the world average. Coal consumption for various purposes in India, China, and the United States is presented in the Fig. 1. The largest coal-producing country in the world is China followed by the United States, whereas India is the third largest coal-producing country.

Rapid urbanization and industrial expansion with the ever increasing population density creates an imbalance in groundwater dynamics in terms of abstraction rate, which is more than the annual recharge rate (Fishman et al., 2011). The utilization of groundwater in the 21st century is sectoral and mostly unregulated and uncontrolled, mainly after the green revolution that started in late 1990s. The green revolution shifted the whole groundwater utilization stress from surface to groundwater and joined hands with industrial development groundwater resources to become a serious concern. In addition to the stress on groundwater quantity, the mining activities affected the quality and quantity of both surface and groundwater resources in many parts of India (Gupta and Nikhil, 2016). The overburdened (OB) dumps, surface impoundments, mine water, industrial effluents, acidic mine drainage and tailing ponds severely deteriorate the natural water environment even long after closure of mine (Gupta and Nikhil, 2016; Kumar and Singh, 2016). The large volumetric discharge of water from aquifers during opencast and underground coal mining operations with toxic elements, and even a small weight proportion of sulphide minerals in coal or mine waste, has the potential to generate acidic conditions and affect the

biosphere (Kumar and Singh, 2016). The present study proposes to review the status of soil quality degradation in the coal mining-affected regions. The evaluation of pollution indices is discussed for determining the contamination level of HMs in coal mine soil. The study also reviews the status of groundwater in the coal mining industry. Coal consumption in developing countries is also discussed in this chapter.

2 Land Degradation Due to Open-cast Coal

Mining Activities

Land is one of the foremost components of the environment that is severely harmed by surface mining activities. Apart from soil pollution, surface mining also led to air and water pollution. Air pollution is caused by the emission of coal dust into the atmosphere, while the leaching of contaminant and acidic mine drainage cause water pollution in the surroundings of the mining area. Increase in respirable particulate matter (RPM/PM₁₀), settleable dust, and suspended particles are the major worries that deteriorate the air quality of the coal mining region. These air pollutants impact human health by causing bronchitis, asthma, and pneumoconiosis.

Coal deposits are located under the dense forest cover, and in India, more than 85% of coal extraction takes place through opencast methods. The opencast coal mine activities cause massive degradation of soil, which, in turn, cause the localized deposit of HMs and minerals. Normally, the mine wasteland comprises loose soil piles, uncovered land, coal mine fire, and OB dumps. The OB dumps are formed by the huge deposition of mine waste at a specific land area, and these dumps are the major source of HMs. The metals present in these dumps cause leaching and also contaminates underground water. Therefore, nowadays, the restoration of OB dumps is a major challenge for environmentalists. Researchers are continuously working to ameliorate OB dumps and remediate or minimize the contamination level. Generally, this is achieved by growing plants that can uptake toxic metals (Ghosh and Maiti, 2020). Hence, the development of vegetation cover is required on OB dumps to minimise pollution level and restore the ecosystem of the affected area.

HM such as Hg is one of the potential elements that lead to serious health risks among mine workers and the population residing in the proximity of coal mines. Table 1 represents the concentration of Hg in the raw and treated coal of major coal mines of India.

According to the reports of UNEP (2014), the removal percentage of Hg) was maximum in the Bina coalfields of Northern Coalfields Limited (NCL),

NCL: Northern Coalfields Limited; CCL: Central Coalfields Limited; JSPL: Jindal Steel and Power Limited.

while the lowest removal percentage was reported in the Gidi coalfield of Central Coalfields Limited (CCL). The difference in the removal percentage of Hg in various coalfields could be due to the higher proportion of HMs in Gidi coalfields than the coalfields of NCL region. The data of three coalfields of CCL is reported in Table 1,

Table 1 Mercury (Hg) concentration in raw and treated coal at different locations in India (*Source*: UNEP, 2014)

<i>Coalfields</i>	<i>Coal types</i>	<i>Hg (ppm)</i>	<i>Removal (%)</i>
Bina (NCL)	Raw	0.093	39.78
	Treated	0.056	
Gidi (CCL)	Raw	0.125	18.4
	Treated	0.102	
JSPL	Raw	0.222	13.06
	Treated	0.193	
Kargali (CCL)	Raw	0.132	25
	Treated	0.099	
Piparwar (CCL)	Raw	0.215	13.95
	Treated	0.185	

which demonstrates the maximum removal percentage of toxic metals in Kargali area.

3 Pollution Load to Water Resources

Scarcity of potable water is apparent in major parts of West Bengal, Jharkhand, Orissa, Uttar Pradesh, Andhra Pradesh, Rajasthan, and Punjab because of anthropogenic/industrial and geogenic pollution (Singh et al., 2012; Tiwari et al., 2016). The surface coal mines affect fresh groundwater resources of the surrounding areas, which eventually have adverse effects on human health and other biotic community (Gupta and Nikhil, 2016). The presence of HMs (Hg, As, Cd, Cr, and Pb) on top or upper soil layer of shallow unconfined aquifers is a serious issue. The HMs leach through the soil, which may be aggravated by the local recharge process of aquifers having high hydraulic conductivity and low vertical hydraulic gradient (Fig. 2). Favourable hydrogeological conditions could be involved in the transportation of contaminants to freshwater reservoirs (Fig. 2).

It is imperative to investigate hydrogeological conditions and periodic monitoring to understand the flow dynamics of contaminants (Gaurav and Khan, 2014; Zhou et al., 2013). In order to understand groundwater flow dynamics, distribution hydraulic heads from the source and flow magnitudes in spatial and various temporal scales could be useful for groundwater resource management. The simulated groundwater contaminant transport has been used in Yishui iron mine, Shandong province, China for effective contaminant management (Zhou et al., 2018). However, little quantitative information is available for transport of coal mine-based contaminants in terms of soil homogeneity, dispersion coefficients, and complex interaction processes. According to a study conducted by Gaurav and Khan (2014), surface and groundwater of the mining areas of Dakra coal mine project, Ranchi, India suffer from high turbidity, presence of high magnesium (Mg), hardness, and

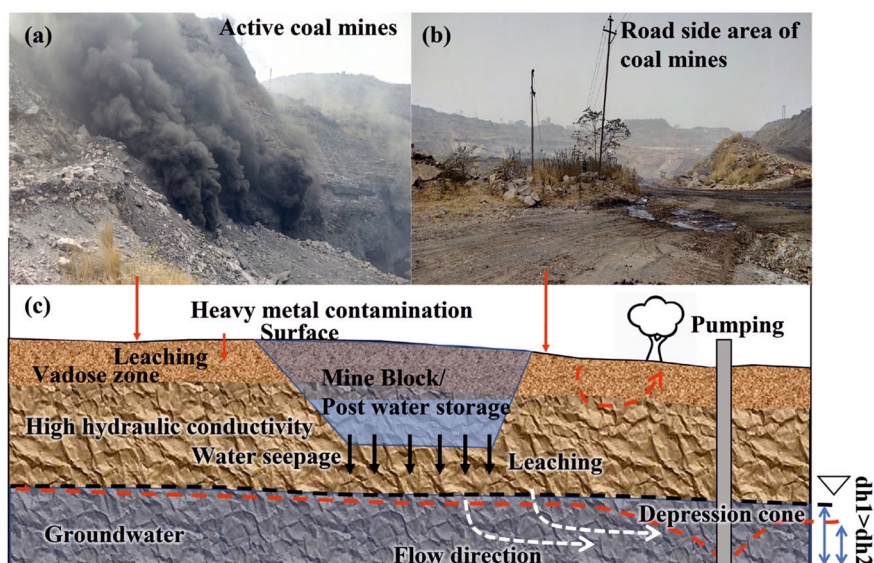


Fig. 2. (a) Active coal mines of Jharia (collected in March 2017); (b) roadside pollution and water seepage from active coal mines of Jharia; (c) schematic representation of open cast mine-mediated soil and water contamination and pollutant pathway to plant. $dh1$ shows hydraulic gradient in normal condition (black dot line), and $dh2$ shows induced pumping (red dot line).

alkalinity above specified limits during monsoon and post-monsoon seasons. The overall presence of earth and alkali earth metals in variable redox conditions, variable pH, total dissolved solids (TDS) and electrical conductivity (EC) are decisive factors which control the groundwater quality for purposeful use and the process of water–rock interaction. The detailed distribution of physical parameters and major groundwater issues of few selected studies on Indian coal mines are presented in Table 2. The open coal mines, having high concentrations of sulphate ions, toxic heavy metals, high biological oxygen demand (BOD; specifically in surface water) and high EC, affect both surface water and groundwater resources (Ugupta and Singh, 2017). Presence of toxic chemicals makes water resources non-supportive for both life and biodiversity.

4 Evaluation of Pollution Load in Coal Mine Areas

The pollution load in coal mine region can be analysed by collecting soil samples from active coal mines. For adequate estimation of pollution load, profile sampling should be conducted to examine the extent of metal concentration. The major benefit of profile sampling is that the movement of HMs is known in different soil profiles. The

Table 2 Groundwater quality and status of few selected coal fields in India

Coalfield	Physical parameters			Major issues	References
	pH	EC ($\mu\text{S/cm}$)	TDS (mg/L)		
Jharia (open cast mine)	6.8-8.3	608-1,350	432-1,080	• TH, TDS, Mg, and SO_4^{2-} are high • Not recommended for drinking	Kumar and Singh (2012)
Jharia	6.2-8.6		437-1,593	• Fe, Mn, Ni, and Pb are above permissible limit • Not recommended for drinking • SO_4^{2-} attributed to pyrite dissolution • Mg, Ca, SO_4^{2-}, and HCO_3^- dominated the water type	Singh et al. (2012)
Jharia coal mines, thermal power plant, and chemical industry	-	-	-	• TDS, Fe, NO_3^-, TH, and EC are above permissible limit • Surface water of this area is more affected than the groundwater • Increasing trend of pollutants towards downstream of thermal power plant	Tiwari and Sinha (2006)
West Bokaro coalfield	-	-	-	• High Fe and Ni content • TDS, SO_4^{2-}, NO_3^-, and TH are above permissible limit • Water is acidic to alkaline in nature • Ca, Na, HCO_3^-, and SO_4^{2-} type water • High SO_4^{2-} concentration attributed to pyrite and gypsum dissolution	Tiwari et al. (2016)
Brajrajnagar coal mine area	4.5-8.9	113-2,094	72-1,340	• Water quality is hard in nature • pH, TDS, and NO_3^- have greater influence on water quality • Ca-, Mg-, and Cl-type water	Sahoo and Khaoash (2020)

(continued)

Table 2 (continued)

Coalfield	Physical parameters			Major issues	References
	pH	EC ($\mu\text{S}/\text{cm}$)	TDS (mg/L)		
Northeast India	-	-	-	• Unusual physicochemical characteristics • High SO_4^{2-} volatilization, vitrinite content, and low ash content • Fe, Mn, Bi, Al, V, Cu, Cd, Ni, Pb, and Mn are abundant	Chabukdhara and Singh (2016)
Damodar Valley coalfield • Jharia • West Bokaro • Raniganj	Jharia: 6.2-8.6 West Bokaro: 4.72-8.43 Raniganj: 7.5-9.1	Jharia: 599-1,653 West Bokaro: -1,598 Raniganj: 409-2,000	Jharia: 459-1,568 West Bokaro: 220-1,502 Raniganj: 336-1,861	• High SO_4^{2-}, Fe, Mn, Cu, and Cr, unsafe for drinking • Water quality of Jharia and West Bokaro is useful for domestic and agricultural use • Raniganj coalfield water is not recommended for domestic and agricultural use	Singh et al. (2009)

TDS: total dissolved solids; EC: electrical conductivity.

collected samples must be analysed in laboratory using standard methods (American Society for Testing and Materials [ASTM] and United States Environmental Protection Agency [USEPA]). If the concentration of some toxic metals is found to be high, then the remediation methods are suggested to minimize their concentration level. Generally, the plant species having higher bioaccumulation capacity are recommended for remediation process (Bortoloti and Baron, 2022). Pollution indices, such as contamination factor, I-geo, pollution load index (PLI), ecological risk index (ERI), and ecological risk factor (ERF), are calculated to know the exact level of soil pollution in coal mining area (Table 3; Ferreira et al., 2022).

Table 3 shows the threshold values of various pollution indices in soil, and the status of pollution in various coal mine soils is evident in Table 4. For calculating pollution indices, the concentration of toxic elements along with the background concentration (reference value) of toxic elements must be known.

I-geo was calculated to explore the pollution level of the coal mine soil of Inner Mongolia, China, and it was discovered that I-geo values of Zn and As were <0 , indicating the safe level, while the I-geo values of Hg and Pb ranged from 0 to 1, indicating safe to moderately polluted level (Table 4; Kou et al., 2022). Different studies have evaluated pollution indices, such as contamination factor and I-geo to

Table 3 Pollution indices and threshold values of soil (Chakraborty et al., 2021; Hakanson, 1980)

<i>Pollution index</i>	<i>Threshold values</i>	<i>Pollution index</i>	<i>Threshold values</i>
Contamination factor (CF)	CF = 0: no contamination	Ecological risk factor (ERF)	ERF < 40: low ecological risk (ER)
	CF = 1: no to medium contamination		40 ≤ ERF < 80: moderately potential ER
	CF = 2: moderate contamination		80 ≤ ERF < 160: considerably potential ER
	CF = 3: moderate to strong contamination		160 ≤ ERF < 320: highly potential ER
	CF = 4: strong pollution		ERF > 320: very high ER
	Cf = 5: strong to very strong pollution	Ecological risk index (ERI)	150 < ERI: low ER
	CF < 6: very strong pollution		150 ≤ ERI < 300: moderate ER
Pollution load index (PLI)	PLI > 1: soil is polluted		300 ≤ ERI < 600: considerable ER
	PLI < 1: soil is not polluted)		ERI > 600: very high ER

Table 4 Pollution indices of various coal mine soils samples

<i>Coalfield</i>	<i>Pollution indices</i>				<i>References</i>
	<i>Geoaccumulation index (I-geo)</i>	<i>Contamination factor (CF)</i>	<i>Ecological risk index (ERI)</i>	<i>Pollution load index (PLI)</i>	
Mpumalanga province, South Africa	As, Cd, Co, Cu, Fe, and Mn: I-geo < 0; Cr, Ni, Pb, and Zn: I-geo > 0	Not reported	ERI < 150	0.89-3.21	Zerizghi et al. (2022)
Inner Mongolia, China	Zn, Ni, and Mn: I-geo < 0	Not reported	Cu, Ni, Pb, Zn, Mn, and As: ERI < 40	Not reported	Kou et al. (2022)
Barapukuria coal mine area, Bangladesh	I-geo: Cu - 0.67, Cr - 1.28, Mn - 2.17, Zn - 0.29, Ni - 0.19, and Fe - 1.93	Not reported	Not reported	PLI: Cu - 2.03, Cr - 3.12, Mn - 1.14, Zn - 1.1, Ni - 3.62, and Fe - 3.2	Hossen et al. (2021)
Ledo, Tinsukia, India	I-geo: Cr - 0.07, Ni - 1.71, Pb - 2.1, and Cd - 1.85	CF: Cr - 1.58, Ni - 4.92, Pb - 6.48, and Cd - 5.3	ERI: Cr - 3.1, Pb - 32.4, and Cd - 159.18	PLI: 4.04	Reza et al. (2015)

assess the contamination level of toxic elements in coal mine soils. For example, in the coal mine soil of Mpumalanga province, South Africa, I-geo was calculated by using the local background value, and the mean I-geo values of soil samples were <0, representing unpolluted level (Zerizghi et al., 2022).

5 Conclusions

The chapter concludes that the open-cast coal mining activities are the major source and cause of land pollution and degradation in the vicinity of coal mines. The land-use soil of these areas is exposed to various HMs such as Hg, As, Cd, Cr, and Pb, which are of severe health risk to the exposed population, especially children. The HM contamination of soil also leads to groundwater and surface water issues of the surrounding mining areas. The extent and load of soil contamination is estimated by using various indices, such as contamination factor, I-geo and PLI. Further, the remedial measures are suggested to minimize pollution load and its impacts. In general, plant species having higher bioaccumulation efficiency are grown on contaminated soils to minimise pollution load.

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Chapter 5

Mining Activities–Risks and Challenges to Environment and Human Health in Sub-Saharan Africa



M. Jeevanandam

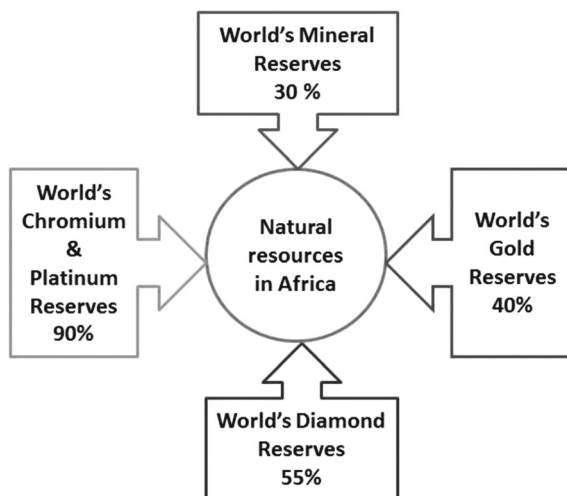
1 Introduction

Mining natural resources in developing countries contribute significantly to economic growth, lifting social status of poor people, providing employment opportunities and alleviation of poverty. In many sub-Saharan African countries, the mining industry makes up for a vital role for its foreign exchange revenue, employment and gross domestic product (GDP). Africa is endowed with 30% of world's total natural resources of metals and minerals (Fig. 1), with vast reserves of strategically important minerals, including platinum group of metals, diamond, gold, cobalt (Co), vanadium, chromite and manganese (Mn), and presently hosting about 700 active mines and more to be planned according to global transition towards low carbon future (Taylor et al., 2009; World Bank, 2017; Standard & Poor Global, 2020).

In sub-Saharan Africa, the mining and extractive industry is dominated by both artisanal miners and multinational companies contributing significant share of the global production of important metals and minerals. However, on the other hand, the overall benefits are opposed by potential negative impacts on the environment and human health that are to be managed properly and effectively to achieve sustainable development. The sustainable development in mining and extractive industry is considered as 'the combination of enhanced socio-economic growth and development, improved environmental protection and pollution prevention' (Hilson and Murck, 2000). The magnitude and impact of mining on the environment and human health trust on the type of mining and the nature of materials involved in the mineral extraction process. Mining areas are generally located in mountainous regions, forests, floodplains, lakes, rivers and coastal areas that are fragile ecosystems harbouring rich and diverse natural resources. These areas are highly sensitive

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Fig. 1 Natural resources distribution in Africa.
(Source: Mining in Africa, 2017, <https://miningafrica.net/>)



and prone to be affected easily under any disturbance. Mining activity is generally through physical and chemical methods that would cause problems to natural resources and land profiles through deforestation, accelerated soil erosion, siltation, and contamination of ground water, acid mine drainage (AMD), loss of biodiversity, dammed rivers and pond waters, waste water disposal issues, natural resource depletion on both land and water, health issues of mine workers, and socio economic issues (Rajaei et al., 2015; Liao et al., 2016; Musa-Obadia et al., 2018; Stewart, 2020). Further, the upsurge in the mining sector leads to the development of infrastructure facilities, such as power lines, rail lines, and roads that affect the migratory routes of birds and animals and promote habitat fragmentation (Haddaway et al., 2019). About 44% of major metal mines in Africa exist inside or within 10 km of a protected area; for example, the Mount Nimba Biosphere Reserve, a world heritage site, in the Republic of Guinea was downsized by 1,550 ha because of iron ore prospecting itself (Edwards et al., 2014). In low- and middle-income countries of the Global South, intense mining and extraction activities enhance water scarcity and threaten species-rich biomes and vulnerable ecosystems (Luckeneder et al., 2021). Numerous studies have been carried out on impact of mining in sub-Saharan African countries related to the environment and human health. Environmental and health effects because of toxic pollutants released due to different mining activities in sub-Saharan Africa are listed in Table 1. Mining industry generates substantial amount of wastes, and waste dumps have significant impact on the surrounding environment. The remains and tailings of mining processes are the most hazardous occupations in the world (Stephens and Ahern, 2001). Leachate from these tailings on water resources, aquatic ecosystem and human health is severe and cause detrimental effects.

Approaches in mining legislations in many developed countries are obligated to ensure and protect the natural ecosystem and to minimise adverse health effects. In

Table 1 Mining impacts on the environment and human health in sub-Saharan Africa

Mining types	Toxic pollutants	Environment impacts	Human health	References
Metal mining	AMD, particulate matter (PM ₁₀ and PM ₀₅), AS, Co, U, Hg, Cd, ~ Mn, Mo, Se, Cr, Pb and CN	Bio- and geo-accumulation of heavy metals, water quality deterioration, deforestation, loss of biodiversity, mine dumps, soil erosion, siltation, loss of wildlife habitat, and land degradation	Oxidative DNA damage, birth defects, helminthic infestations, psychotic disorders, musculoskeletal problems, mercury and cyanide intoxication, respiratory distress, memory loss, tremors	Isidro et al. (2017), Machacek (2020), Banza Lubaba Nkulu et al. (2018), Mkandawire et al. (2017), Abigail et al. (2021), Ouma et al. (2022)
Coal mining	AMD, particulate matter (PM ₁₀ and PM ₀₅), SO _x , NO _x , O ₃ , benzene, H ₂ S, Co, Cr, Ni and Pb "	Air and noise pollution, aquifer deterioration, agricultural soil contamination with heavy metals, crop reduction, river and lake water contamination because of AMD and landscape destruction	Pneumoconiosis (black lung disease), chronic obstructive pulmonary disease (COPD), chronic bronchitis, progressive massive fibrosis (PMF) and silicosis among coal miners; asthma, chest coughs and phlegm	Markandya and Wilkinson (2007), NIOSH (2011), Robert et al. (2021), Maya et al. (2015)
Sand and gravel mining	Fine silica dust, oil spills and grease	Loss of benthic organisms and species diversity, depletion of fish production, increased water turbidity, changes in river morphology, deforestation, collapse of river bank, dust, noise and air pollution	Silicosis and cancer, malaria and schistosomiasis	Davies (2012), Syah and Purnaweni (2018)

addition, the mining legislation urges mining companies to adopt proper rehabilitation plans to restore former mining sites into safe environmental conditions after lifetime of the mine is over. However, this is not always the case in many developing countries, especially in sub-Saharan Africa. Lack of proper environmental governance and detailed assessment of mining activities lead to severe impingement ecosystems as well as human health (United Nations Educational Scientific and Cultural Organization [UNESCO], 2014–2015).

2 Mining Regime in Africa

Mining activities in sub-Sahara African countries could substantially influence the natural environment, and its impact potentially depends on the scale of operation and the infrastructure associated with it. The mining activities can affect the environment either directly, which occurs in the vicinity of the mine by removing and degrading natural habitat, or indirectly by a consequence of contamination, infrastructure facilities and migration of people. Mining sector in Africa can be classified as formal and informal. The formal sector operates legally, which comprises either registered group of people (e.g. artisanal miners association) or enterprises owned by individuals. The informal sector comprises a group of miners or individuals who operates illegally and are not controlled by government (Ledwaba and Nhlengetwa, 2016; Mhangara et al., 2020). Although artisanal and small-scale mining (ASM) are mostly informal and illegal activities in sub-Saharan Africa, ASM is one of the most important non-agricultural activities for livelihood and 60 million people rely on this to extract minerals (World Bank, 2019; Machacek, 2020).

Economically valuable commodities, such as tin, cobalt, lithium, gold, diamonds, tantalum and rare earth elements, and other less valuable resources, such as sand, coal and mica, are the products of ASM activities (Schwartz et al., 2021). In Zimbabwe, gold is a primary export product, and recorded 63% of gold is produced by artisanal mining sector, which involves as many as 1.5 million people (London School of Economics [LSE], 2021). However, owing to rudimentary techniques and tools involved in ASM methods, environmental consequences and occupational health hazards are diverse and complex.

3 ASM Effects on the Environment and Health

Although ASM contributes significantly to economic growth, employment generation and poverty alleviation, poor legislations and lack of proper governmental control on ASM result in worse environmental and health consequences than large-scale mines in sub-Saharan African countries. During life cycle of ASM, significant environmental deterioration is inflicted to the physical and biophysical environment through mineral exploration, exploitation and processing activities. The springing up of new settlements around mining areas and discharging hazardous wastes from mines without proper disposal systems have placed both human health and ecosystems at risk. The serious environmental issues related to ASM are land cover change, deforestation, siltation in streams and dams, soil and water contamination, soil erosion, and mercury (Hg) and cyanide contamination (Lobo et al., 2015; Isidro et al., 2017; Machacek, 2020). Innumerable health consequences because of ASM, such as exposure to toxic chemicals (cyanide, mercury, zinc vapour and other acids), dust inhalation (tiger's-eye pneumoconiosis), accidents, illicit drug, alcohol

addiction and violence, have been reported (World Health Organization [WHO], 2017).

The recent quantitative estimation of environmental effects because of ASM throughout African continent has demonstrated a considerable deforestation of tropics and vegetation and health degradation, which may lead to climate change and future droughts (Bagley et al., 2014; Girard et al., 2022). Machacek (2019) has established typology of environmental effects because of ASM in the Great Lakes Region of Africa (Rwanda, Burundi, Uganda, Tanzania, Kenya, and the eastern part of the Democratic Republic of Congo [DRC]) based on primary research, comparison with foreign approaches, and interview with experts. These environmental effects are grouped into the following four categories: (i) changes in landscape structure–deforestation (primary and secondary) and change in land cover; (ii) influence of geomorphological processes (weathering, mass movements, fluvial process, Aeolian process, and creation of new anthropogenic forms); (iii) influence of hydrological regime (water contamination, sedimentation of water stream); and (iv) influence on fertility of soil (soil contamination, high dustiness, change in land use). Mining and extraction of metals by ASM has posed grievous environmental damage and health hazards through usage of toxic chemicals, such as mercury, cyanide and other toxic elements, and release of radioactive elements into the environment which is associated with ores. Tin (cassiterite, tin oxide mineral [SnO_2]) exploration and exploitation in the Jos Plateau of Nigeria has caused destruction of pastoral land, soil degradation through soil erosion, loss of soil nutrients and organic matter, and damage to crops. Radioactive wastes, associated with tin ore, from mine tailing and mine ponds were released into the environment, causing serious health hazards among local communities (Adeyinka and Ako, 2019).

4 Metal Mining Impacts

4.1 Gold Mining Impacts

The growing demand of metal commodities globally has led to increased production of metal ores through new mining plans, expansion of existing mines and enhancing the extraction processes (Luckeneder et al., 2021). Africa's gold mining industry itself contributes 60% of the total mineral production, which accounts for 22% of the earth's total gold (Mining in Africa, 2021). Half of the world's platinum group of metals, such as gold (36%) and cobalt (26%), are found in South Africa (SA) (Ouma et al., 2022).

In sub-Saharan Africa, ASM is often associated with gold mining, where mercury usage results in environmental degradation. Mercury is frequently used for amalgamation process to extract fine gold from ores, as it is the cheapest and fastest technique. For every 1 kg of gold produced, about 1.32 kg of mercury is used, which in inorganic form is directly transferred into the soil and aquatic environment

(Matshusa et al., 2012). Owing to its volatile nature, mercury remains suspended in air during mining and roasting process, and is inhaled by humans to cause intense health impairments (Fashola et al., 2016). Even after the Minamata Convention of 2017, comprehensive efforts to eradicate or minimize the usage of mercury have failed and it remains the first choice to extract gold from ores in ASM (United Nations [UN], 2022; Anna et al., 2022). Inorganic mercury derived from ASM processes is converted into organic methyl mercury (MeHg) by microbes present in the aquatic environment and accumulates in food chains, such as fish, mollusks and crustaceans. Ingestion of MeHg is primarily harmful to the central nervous system and causes nerve and brain damage (Reichelt-Brushett et al., 2017). Mercury vapours produced from the roasting process of gold–mercury amalgam leads to acute mercury exposure, which may cause respiratory distress, memory loss, tremors and even death (Bose-O'Reilly et al., 2016; Esdaile and Chalker, 2018).

According to Mercury Watch (2017), Burkina Faso (production 35 tonnes/per year) in western Africa is ranked fourth in mercury emissions because of ASM of gold processes. According to Porgo and Gokyay (2017), the study conducted in the Essakane gold mining site of Burkina Faso showed that artisanal gold miners and industrial open pit mining activities have caused intense air pollution, natural landscape degradation, and loss of agricultural and grazing lands.

Tanzania is one of the global 'mercury hot spots' of elemental concentrations of mercury or MeHg in the environment and humans (Evers et al., 2014). Contaminated water after amalgamation process is directly discharged into the Lupa river without treatment of mercury, thus leading to environmental impingement of Lake Rukwa and health hazards among the populations of both Matundasi and Makongolosi regions of South Tanzania. In these regions, the mercury concentration in human hair significantly exceeds the US Environmental Protection Agency's limit of 1.0 parts per million (ppm) (Intergovernmental Forum on Mining [IGF], 2017).

The artisanal gold mining in the Mzingwane district of Zimbabwe has caused life-threatening environmental damages, which include veldfire, considered as a serious problem, followed by siltation in local dams, deforestation and land degradation. Even the animals of the region have suffered from burns and deaths because of veldfires caused by local gold panners. Everyday, around 172,000 kg of soil is removed by artisanal panners and processed in the Umzingwane river and eventually is carried by streams into dams, resulting in dire siltation. Owing to the process of gold amalgamation, mercury has also been reported in Mtshabezi dam, south of Esigodini in the Mzingwane district of Zimbabwe, with significant impact on wildlife, domestic animals and humans (Ncube-Phiri et al., 2015).

The results of the conducted physicochemical studies of soil in southwestern Nigeria indicated that there was grievous soil degradation, heavy metal contamination and loss of important soil nutrients because of the artisanal gold mining activities (Adebayo et al., 2017). Another study carried out by Ralph et al. (2018) in Batouri, the gold district of East Cameroon, demonstrated that gold miners were exposed to high risks of diarrheal conditions, respiratory conditions, dermatophyte infections, helminthic infestations, psychotic disorders and musculoskeletal difficulties because of chronic intoxication of mercury.

Cyanidation is another harmful process used to extract gold from low-grade ores (0.5 g/tonne of gold) (Deschenes, 2016). Free cyanide (HCN, CN^-), main tailings associated with gold mining, is easily released in the environment, and even at low concentration poses risks to human health (González et. al., 2022). Prolonged exposure to small amounts of hydrogen cyanide (6–10 mgm) could cause chest pain, breathing difficulties, changes in blood, and enlarged thyroid gland (Agency for Toxic Substances and Disease Registry [ATSDR], 2006).

4.2 Cobalt Mining Impacts

In order to mitigate global warming and climate crisis because of greenhouse gas emissions in the present era, the usage of lithium–iron batteries and electric vehicles is being promoted globally. Cobalt is an essential part of lithium–iron batteries, electric vehicles and cell phones. In the last decade, the global demand of cobalt has tripled and is going to increase further by 2035 because of decarbonisation efforts (Filipe et al., 2021; Faraday Institute, 2020). Hence, cobalt mining has increased with profound negative environmental consequences. Cobalt enters in the environment through soil, rocks, air dust, water plants and radioactivity at mining areas. Cobalt cannot be destroyed once it enters the environment. Radioactive emissions and discharge of cancer-causing particles and detrimental elements, which may cause asthma, pneumonia, nausea, heart diseases and thyroid damage, are associated with cobalt mining activities (Fordyce, 2013; Farjana et al., 2019). DRC, the world's largest cobalt producer, contributes approximately 60% of global cobalt production from Katanga copper (Cu) belt, with substantial portion extracted by artisanal miners (Banza Lubaba Nkulu et al., 2018).

A study conducted in Kolwezi town situated in the western Katanga province of DRC (Banza Lubaba Nkulu et al., 2018) described that the surface dust generated due to artisanal mining of cobalt contained 1,100- $\mu\text{g/g}$ cobalt and 2- $\mu\text{g/g}$ uranium, an associated ore mineral. Prolonged exposure to these surface mineral dusts caused high concentration of urinary cobalt in adults (64 $\mu\text{g/g}$), children (193 $\mu\text{g/g}$) and miners (133 $\mu\text{g/g}$). In addition, oxidative damage to deoxyribonucleic acid (DNA) was discovered in highly-exposed children that could lead to increased risk of cancer.

4.3 Copper Mining Impacts

Copper and its associated alloys are inevitable and essential part of the construction sector, telecommunication industry, electric and electronic products and transportation equipments. According to LeCain (2009), copper mining results in the 'mass destruction' of the environment. DRC, Zambia, South Africa and Namibia are leading copper exporting countries in sub-Saharan Africa. The Central African Copper belt, located between the border regions of Zambia and DRC and known for its vast copper

as well as zinc, cobalt and gold deposits, has witnessed intense mining activities. This region is also a habitat for wildlife, such as large mammals; however, mining expansion in this zone has caused serious environmental and health impingements because of contamination of soil, water and air, forest clearings, and effects of population pull factors of mining towns (Mwitwa et al., 2012; Pesa, 2022; Luckeneder et al., 2021).

The aquatic biota and human health are dangerously affected at the Kafue river basin (743,000 km²), a copper belt province, of Zambia due to mining waste effluents (Kapungwe, 2013). The water and sediments of the Kafue river are primarily contaminated by copper, cobalt, chromium (Cr), lead (Pb) and cadmium (Cd) (Ikenaka et al., 2010). According to Mkandawire et al. (2017), the heavy metal assessment conducted in the Kafue river indicated a heavy enrichment of copper near copper belt mining areas, with high evaluates of arsenic (As), manganese and cobalt. The Kafue flatbeds, 6,500-km² floodplain along the river, serve as a sink of heavy metal species. These wetlands or floodplains absorb significant amount of contaminants, thus displaying a serious threat to the wildlife, environment and human health. The wildlife, such as kafuelechwé (antelope), Tilapia and African catfish and Echinochloa (grass), has been observed as bearing high levels of lead (mg/g of dry weight), thus exhibiting risk to higher tropic organisms (Mkandawire et al., 2017; Ouma et al., 2022).

4.4 Coal Mining Impacts

Coal has played a decisive role in the Industrial Revolution, and is regarded as the world's most prominent source of solid fuel (Miller, 2011a). Coal fulfills world's electricity needs (38%) and accounts for 44% of global CO₂ emissions, thus becoming a major focus of climate change (International Energy Agency [IEA], 2018, 2019). The coal extraction process and waste disposal affect both directly and indirectly the soil, air, water, ecosystems, and human and animal health. The most important environmental fears are climate change and biodiversity (United Nations Environment Programme [UNEP], 2019). Many of the coal processing activities in the coal lifecycle have caused serious damages to the environment in the form of land subsidence, AMD, water pollution, destruction of the wildlife habitat and ecosystems, cutting of forests, sedimentation, and erosion (Dai et al., 2017; Giam et al., 2018; Robert et al., 2021). Water is crucially associated with coal extraction process, and hydrological consequences of surface and underground coal mining processes are very complex and significant. Aquifer deterioration, lowering of water table, destruction of natural drainage pattern, sand surface and groundwater contamination are the serious consequences of coal mining activities (Yang et al., 2018; Masood et al., 2020). Fugitive dust, generated by various coal mining activities, such as extraction, cleaning, transportation, and storage, presents a serious risk to the surrounding environment and human health (Miller, 2011b). The occupational health consequences of the dust generated by coal extraction process causes pulmonary diseases, such as coal workers' pneumoconiosis ('black lung disease'), chronic obstructive pulmonary

disease (COPD), chronic bronchitis, progressive massive fibrosis (PMF) and silicosis among coal miners (Markandya and Wilkinson, 2007; National Institute for Occupational Safety and Health [NIOSH], 2011; Robert et al., 2021).

South Africa, a leading producer of coal among sub-Saharan African countries, employs 90,000 miners and is ranked fifth in global coal production (Elektrisiteit Svoorsienings Kommissie [Electricity Supply Commission/ESKOM], 2016; Olufemi et al., 2019). This has caused unprecedented environmental degradation, and according to nonprofit repository, GlobalGiving Atlas, South Africa has become the 12th largest global producer of greenhouse emissions. The eastern part of Johannesburg in South Africa is listed as one of the world's most polluted areas. South Africa's coal mining areas are covered with coal ash, smog and stink of sulphur. A study showed that more than 5,000 South Africans die annually in the nation's coal belt due to atmospheric pollution (Dailymaverick, 2021).

The mining methods widely utilized in South Africa are surface mining (strip mining) and underground mining (room and pillar mining) (Akinlabi, 2019). In strip mining, the overburden (vegetation, soil and rocks) is removed to reach coal seams and dumped. In room and pillar mining, coal seams are mined between pillars, which remain intact to support the roof. Strip mining destroys the landscape and decreases the aesthetic value of the natural environment. On the other hand, waste products generated by room and pillar mining cause air and water quality deterioration by coming in contact with the atmosphere.

Emalahleni, a coal municipality in the Mpumalanga province (Fig. 2) of South Africa, with maximum number of coal mines with approximately 22 operations around the area (Benchmarks Foundation, 2017), is intensely polluted with sulphur dioxide (SO_x), nitrogen oxides (NO_x), ozone (O₃), benzene, hydrogen sulfide (H₂S) and particulate matter (PM₁₀ and PM_{2.5}) (Botha, 2014). Emalahleni municipality has been declared as having the worst air quality with the dirtiest atmosphere in the world (Maya et al., 2015; Munnik et al., 2010). Children in Emalahleni town are exposed to polluted toxic air in schools and homes, causing serious health consequences, such as bronchitis (16.5%), asthma (7.1%), chest cough (10.1%), and phlegm (25%) (Albers et al., 2015; Nkosi et al., 2017). The quality of agricultural soil in Emalahleni province, the largest producer and supplier of agro products in South Africa (Bureau for Food and Agricultural Policy, 2012), is polluted with atmospheric deposition of heavy metals, such as cobalt, chromium, nickel (Ni) and lead. This has resulted in soil acidity, worsened crop quality, and reduced agricultural produce, thus reportedly increased food prices (Bureau for Food and Agricultural Policy, 2012; Maya et al., 2015).

In Nigeria, abandoned coal mines, being reworked by local artisanal miners without proper safety measures, have caused deaths in Zamfara state because of lead poisoning (Ojonimi et al., 2021).

Fig. 2 A woman digs for coal at an abandoned coal mine in Wesselton on the northern outskirts of Ermelo, Mpumalanga, South Africa. (Source: <https://african.business/wp-content/uploads/2022/07/mpumalanga-coal>)



5 Effects of Mine Tailings and Acid Mine Drainage

The remains and tailings of mining processes have become the most hazardous and emerging issues in mining milieu, leading to life-threatening environmental degradation and short- and long-term health consequences. Toxic metals from hazardous mining wastes leached out in the environment and their non-degradative properties make them persistent causes of long-term serious manifestations on ecosystem and human health (Singh et al., 2011; Oyewo et al., 2018). Mine tailings containing harmful metals and metalloids (Zn^{2+} , Ni^{2+} , Pb^{2+} , As^{2+} , and Cu^{2+}) are the point source of pollution to the nearby environment. The leaching of these metals along with sulphate ions from mine wastes perpetually engenders pollution risk to the aquatic system, such as groundwater and surface water, at nearby dumping locations (Oluranti et al., 2020).

The ASM gold tailings because of amalgamation process contain substantial amount of mercury. Difficult to mitigate, the suspended mercury travels for hundreds of kilometers in river waters and contaminates the soil and aquatic environment (Diringer et al., 2015). Significant loss of mercury takes place as these tailings may contain 50–5,000-mg mercury per kilogram of ore (Cordy et al., 2011). Mercury in the aquatic environment may lead to mercury methylation by bacteria and bioaccumulation of MeHg in fish species (Veiga et al., 2014). Bose-O'Reilly et al. (2016) documented mercury uptake by rice paddies because of mercury-contaminated water used for irrigation, and as high as 1.2 ppm mercury was observed in edible grain, that is more than 10 times the recommended limit set by WHO. Owing to polluted air because of dumping of gold mine tailings, the local communities living up to a distance of 5 km (Benchmarks Foundation, 2017) are always at risk of acquiring respiratory clinical manifestations, such as chronic bronchitis, chronic cough, emphysema and pneumonia (Nkosi et al., 2015).

Owing to mining activities for several decades, the Gauteng province of South Africa is characterized by the widespread dumping of mine tailings. These dumps, with high concentrations of arsenic, lead and aluminium (Al), have unceasingly

confronted negative consequences on the surrounding environment through contamination of soil and water. According to Mpanza et al. (2020), local communities living in the vicinity of gold mine villages of Gauteng Province in South Africa suffer from cancer, asthma, and eczema, and several deaths have been reported due to tuberculosis.

The Gruben river in Namibia is under severe stress during its episodic flow because of spoil heaps of copper mining that releases 94.7% of copper (36 mg L^{-1}), 90.5% nickel (35 mg L^{-1}) and high concentrations of copper in suspended sediments ($10,500 \text{ mg L}^{-1}$) (Taylor and Kesterton, 2002; Ouma et al., 2022).

Acid mine drainage, a consequence of coal and metal mining activities and abandoned mines, has become an emerging global issue. AMD is generated by sulfide-bearing ore minerals, especially pyrite (FeS_2) or pyrrhotite (FeS), and, exposed to oxygen and water, forms sulphuric acid by the process of hydrolysis and oxidation. Coal mine tailings contain sulfides and other metalloids, which, coming in contact with surface and groundwater in the presence of air, can produce a combination of low-pH (acidity) metal contamination in the aquatic environment. AMD could have severe toxicological effects on aquatic ecosystems, such as rivers and lakes, because of frequent loading with high metal concentration around active and abandoned mines (Iwasaki et al., 2020). The toxic aquatic environment may immediately kill marine organisms, or prolonged exposure with low concentration of contamination may cause severe effects, such as mortality, infertility, growth retardation, lesions and deformities (Ministry of Mines and Steel Development [MMSD], 2014).

Issues of AMD in South Africa originated from abandoned mines, particularly from underground mines after the closure of large scale mines in the 1970s (Makgae, 2012; Feris and Kotze, 2014). AMD issuances from abandoned mines in South Africa occurred in Mpumalanga coal fields, KwaZulu-Natal coal fields, KOSH gold field, Free State gold field, Evander gold field, South Rand gold field, O’Kiep copper district, Eastern and Western basins, and Far Western and Central basins (Council for Geoscience, 2010). The Yellow Jacket and Mazowe rivers in Zimbabwe are contaminated with heavy metals, acidity and habitat loss because of Iron Duke pyrite mine, which contains ferrous alloys (cobalt and nickel), non-ferrous metals (copper, lead and zinc) and sulphahtes (Ravengai et al., 2007; Tazvivinga et al., 2012; Ouma et al., 2022). The formation of AMD and cyanide and heavy metal contamination in the Zambezi Basin of Zimbabwe have occurred due to ore geochemistry, mineral processing activities, waste rocks and slimes, which negatively affected aquatic biota communities, such as microbes, macro-invertebrates, fish and macrophytes (Love et al., 2006; Huang et al., 2019; Ouma et al., 2022).

6 Impacts of Sand and Gravel Mining

Global economy is built on natural aggregates such as sand and gravel. In order to meet the growing demand of the infrastructure and construction industry, about 50 billion tonnes of sand and gravel are excavated every year (World Wide Fund for

Nature [WWF], 2018). The sand and gravel mining industry accounted for 85% of global mineral exploration (Leal Filho et al., 2021; Institute for Security Studies [ISS], 2020). These valuable and non-renewable commodities make up for a crucial role in the rapidly expanding African construction industry. According to Bendixen (2020), sand requirement in African countries would increase drastically in the next 10 years and could face terrifying extraction practices in the future. However, because of crude mining methods using heavy machines and unsustainable sand mining constitute complexities around aquatic ecosystems, such as streams, rivers, floodplains, deltas, coasts, lagoons and estuary (Esguerra et al., 2008; Bhattacharya et al., 2019). Natural aggregates, particularly sand, serve as principal source of construction globally. Rivers and floodplains are valuable sources for well-graded sand. The potential environmental degradation in exploitation of natural aggregates depends on the type of mining involved in the riverine environment. Mining activities for riverine aggregates are classified into the following three categories: scalping or skimming, in-stream or wet mining, and dry mining. In-stream mining dominates in aggregate extraction, which involves direct removal of substratum of a flowing stream or river, followed by dry mining and scalping or skimming, which, respectively, consist of mining in floodplains or in ephemeral streams and removal of top layer of the exposed sand bar (Padmalal and Maya, 2014). In-stream or wet mining leads to changes in channel bed and bank morphology, and water quality deterioration (Molenda et al., 2021). The biological environment, such as species diversity, benthic organisms, and vegetation pattern of river or stream, is severely affected by the removal of substratum (Yen and Rohasliney, 2013; Ako et al., 2014; Pitchaiah, 2017). Sand mining in floodplains causes change in land use and decline of water table. The other potential negative environmental impacts due to sand mining activities include deforestation, habitat destruction, functionality of river ecosystem and natural hydrological process (Saviour, 2012; Jordan et al., 2019; Koehnken et al., 2020).

On the other hand, gravel mining through quarries in mountains and hills has resulted in serious soil and land degradation by removal of top soil and trees and loss of agricultural and grazing land. Soil nutrients deficiency and crop reduction have been reported in many gravel mining areas. Landscape destruction, a severe consequence of gravel mining, resulted from excavated pits and trenches, which decrease the aesthetic value of physical environment, makes land unsuitable for any other productive purpose (Julie et al., 2018; Endalew, 2019; Belay et al., 2020; Gashaw and Aklilu, 2020). The potential environmental impacts and occupational health hazards because of sand and gravel mining in the selected sub-Sahara African countries are given in Table 2.

Table 2 Environmental impacts and occupational health hazards because of sand and gravel mining in sub-Saharan Africa

Country	Location	Type of material extracted	Source	Environmental impacts
South Africa	Nzhelele Valley, Limpopo Province	Sand and gravel	Stream and river, terraces, floodplain and channels	Destruction of fisheries and recreation of river channel instability (Mathada and Kori, 2012)
Tanzania	Zanzibar	Sand	Coastal areas	Deforestation (mangroves), loss of native trees (coconut), deaths (quick sand) (Ladlow, 2015)
Nigeria	(1) Luku, Niger state	Sand and gravel	Inland, river banks and channels	Deforestation, farm and grazing land reduction, collapse of river bank, landscape destruction, water and air pollution (Ako et al. 2014)
	(2) Ado Ekiti metropolis	Sand mining	Inland, river mouth and banks	Loss of agricultural land and vegetation, water and air pollution (Atejiye and Odeyemi, 2018)
	(3) Lagos	Sand mining	Coastal areas	Dust, road destruction, land degradation, flooding, erosion, water pollution and public health (Aliu et al., 2022)
Benin (West Africa)	Eastern and Western Complex	Sand mining	Coastal lagoon, river channels, delta	Destruction of aquatic ecosystem and organisms (Lalèyè et al., 2019)
Ethiopia	(1) Addis Ababa	Gravel quarrying	Hills	Dust, noise pollution, surface and groundwater pollution, destruction of heritage and landscape (Semere, 2013)

7 Conclusions

Mining activities in sub-Sahara African countries have substantially influenced the natural environment and human health. Both formal (legal) and informal (illegal) mining activities in sub-Sahara African countries have made up for a vital role for employment and revenue generation. Economically important and valuable minerals and metals are mined by ASM methods in sub-Sahara African countries. However, because of rudimentary techniques and tools involved in ASM methods, the environmental impacts and occupational health hazards are diverse and complex. Poor legislations and lack of scientific knowledge of ASM methods lead to serious environmental deterioration and health consequences in spite of its significant contribution to employment, economic growth and poverty alleviation. Recent research conducted on ASM methods throughout the African continent showed that there was a substantial amount of deforestation in the tropics and vegetation and health degradation, which could lead to climate change and future droughts. Mine tailings and AMD

have become emerging issues in many African countries because of leaching and discharge of toxic metals into aquatic ecosystem, thus constituting long-term serious effects on ecosystem and human health. Natural aggregates, such as sand and gravel, are valuable and non-renewable commodities of the African construction industry. Requirement of both sand and gravel in African countries has increased drastically, eventually leading to rude and unsustainable mining methods using heavy machinery. The biological environment, such as benthic organisms, diverse species, and vegetation pattern of rivers, floodplains, deltas, coasts, lagoons, estuaries and streams, is seriously affected by sand mining activities around aquatic ecosystems. On the other hand, gravel mining in mountains and hills through quarries has resulted in soil and land degradation because of soil and vegetation destruction.

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Chapter 6

Mining Impact on Uttarakhand's Hill Ecology



Soniya and Anurag Linda

1 Introduction

The mining industry regulation is vital for our economy because it provides raw materials, minerals, and metals. It is vital for producing goods and services, infrastructure, and advancing the quality of human lives in the society (Agboola et al., 2020). Metals are fundamental components of life and are part of the food chain; Copper (Cu) and Zinc (Zn) are some of them that are essential to life (Allan, 1997). With the advent of human civilization, the refined metal was discovered around 11,000 years ago (Haque et al., 2014; Lechner et al., 2017). Mining of mineral ores is done to obtain metals, obsolete technologies of exploring minerals have been prevalent since centuries. The metal age began with sharp cutting objects like axes and chisels, the evidences can be traced to the pre-Harappan and pre-Christian eras. The metallic and nonmetallic ores are generally mined from the earth and the process of obtaining the ores and metals from the earth is known as mining. Ores are generally present naturally in lodes, veins, seams, reefs, and placer deposits. Mining is carried out to extract different metals, fossil fuels, precious stones, and nonmetallic ores like carbonate rocks, rock salt, potassium, gravel, clay etc (Evans, 2009). The mining procedure is necessary for extracting any mineral since most of the minerals are not freely available in the crustal rocks.

The present need of different mining industries is to adopt different sustainable mining practices. Sustainable mining is based on the suggestions given by

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the United Nations, World Commission of Environment and Developments Landmark our Common Future Report. A sustainable mining framework focuses on minimizing the impact of mining on the environment; the strategies include limiting land disturbance, pollution reduction, automation, renewable energy consumption, and proper closure and reclamation of mined lands.

1.1 History

Different types of mining have been practiced for centuries and the techniques practiced were different in different civilizations and cultures. It developed gradually through different ages and finally evolved into a science of exploration from the skills of past practices (Ruvelta, 2017). Since ancient times, mining has played a key role in the history of human beings and it is still persistent. According to historical records, 'mining' and 'mineral processing' were not considered independent activities in the past, there was a bare minimum tools, infrastructure, and expertise, but with great resourcefulness, our ancestors dug minerals from the earth's crust (Eckelman and Graedel, 2007). The mining technique in different countries is unique and varies with cultural and regional diversity. Different workers have discussed some interesting aspects of mining and metallurgical history (Gregory, 1980; Lacy and Lacy, 1992). The process and type of mining depend upon the occurrence of mineral resources and the ease of extraction with the available technologies and techniques.

1.2 Types of Mining

Based on the occurrence and distribution of different minerals, different types of mining are discussed below:-

1.2.1 Surface Mining

Surface mining is suitable for extracting minerals near the earth's surface or in the upper crust. Compared to underground mining, it is usually the most cost-effective and easy. Surface mining excavates various rock and mineral ores like slate, marble, bauxite, hematite, magnetite, chalcopyrite, etc.

1.2.2 Underground Mining

Underground mining is suitable for extracting minerals deep beneath the earth's surface at the lower crust. Gold and silver are the most common minerals recovered through underground mining.

1.2.3 Shaft Mining

Shaft mining is a category of underground mining that includes deep underground mining using an array of shafts. This method excavates minerals and ores by pushing shafts vertically downwards, also known as shaft sinking. It is excellent for concentrated materials like iron, coal, and other minerals deep beneath the earth's surface.

1.2.4 Open-pit Mining

Open-cast mining is a type of mining in which overlying materials are removed to make a big pit, and finally, materials are excavated from these open pits. It is carried out when the overlying material is loose and unsuitable for mining or there is a massive deposit of rock and minerals on the earth's surface under a thin cover of earth material.

1.2.5 Longwall Mining

Longwall mining is underground mining in which a slice of rock or mineral deposit is excavated. When minerals are extracted, steep walls are formed, which can run for 3 to 5 kilometers in the case of coal.

1.2.6 Room and Pillar Mining

Room and pillar mining is a type in which rocks and minerals are extracted from a horizontal plane; horizontal arrays of rooms and pillars are formed during this technique. When removed, ores form "rooms," leaving "pillars" of undisturbed material to hold the roof overburden.

1.2.7 Hydraulic Mining

Hydraulic mining involves dislodging rock material or moving mud with high-pressure water jets. It is generally used to extract minerals like gold, tin, clay, and coal.

2 Mining in the Himalayas

India has rich mineral resources and a wide range of landforms from east to west (Fig. 1). The climate varies from the mountains region of the snow-capped Himalayas to the coastal zone of peninsular India from north to south. It has deserts in the west and dense forests in the east. These different climatic regimes show rich biodiversity.

Himalaya is the abode of more than 9,000 glaciers; these glaciers play an important role in the local hydrological cycle. In recent decades, due to the impact of climate change, more of the Himalayan glaciers are melting, giving rise to glacial lake formation and increased glacial melt runoff. Population pressure in the Himalayas is also resulting in deforestation, the combined effect of land use and land cover change due to human perturbances, and climate change has changed the energy budget of the valleys, leading to cloud bursts and increased rates of landslides. Events such as mining, tunneling, and encroachment of natural drainage patterns are further making the fragile ecosystem of the Himalayas more prone to associated disasters and environmental degradation (Fig. 2). Thus, mining different natural resources from the Himalayan region needs different management strategies, depending upon the stability of the slope, prevailing climate, geotectonics, and the extent of vulnerable society.



Fig. 1 Map showing Himalayan States from East to West.



Fig. 2 Slate mining reserve in Himachal Himalaya.

Indian Himalaya is divided from south to north into the sub-Himalaya or Shiwaliks or Siwaliks (up to 1000m amsl), lesser Himalaya or middle range Himalaya (2000–3000m amsl), great Himalaya and trans-Himalaya (> 3000m amsl). From west to east, the Himalayas in the Indian territory are sub-divided into different regions and are categorized as follows:

- J&K Himalaya or the northwestern Himalaya in the state of Jammu and Kashmir
- Shimla Himalaya or HP Himalaya in the state of Himachal Pradesh (HP)
- Garhwal Himalaya extending in the hill districts of Garhwal (previously referred to as UP Himalaya)
- Kumaon Himalaya in the Kumaon division of Uttarakhand.
- Northeastern Himalaya extends to Arunachal Pradesh, Sikkim and Assam.

3 Mining and its Impact on Environment and Ecology

Mining activities create a lot of ecological imbalances, including the destruction of wildlife and habitat. It is widely acknowledged that mining devastates the local ecosystem; it causes land degradation, LUCC (Land Use and Cover Change), deforestation, biodiversity loss, water system impairment, and livelihood impacts, among other things.

The environmental effects of mining have been extensively researched and documented. Mining has many environmental consequences on a local, regional, and global scale; it has a direct or indirect effect on a regional and global scale (Sonter et al., 2018). Mining activities cause numerous ecological imbalances, including habitat degradation (Yeganeh, 2020). Mining processes also alter the natural climatic

conditions due to carbon emissions and adversely affect Human health, vegetation, and Biodiversity.

3.1 Ecological Impact of Mining

3.1.1 Air Pollution

Mining releases toxic dust and gases hazardous to miners and the natural environment. Exposure to dust during mining operations can cause various types of respiratory and other health-related diseases; various studies have reported that mining could instigate the formation of scar tissues in the lungs over time (Petsonk et al., 2013).

Diesel and gasoline machinery emit Particulate Matter (PM), NO_x, SO_x, and carbonaceous contents, both organic and inorganic; these pollutants degrade the local air quality of a region and also amplify the process of global warming (Kumar et al., 2016). High dust load is generated during open cast mining, one of the biggest culprits for PM generation in the mining area. PM emissions from mining operations could penetrate deeper into the lungs and bronchial tubes, resulting in the aggrandizing of respiratory disease cases among the mining personnel and the residents. The adverse impact of toxic dust from mining on the region's flora and fauna is also reported in numerous studies (National Research Council, 2006) and is discussed in the subsequent sections.

The usual benchmark for setting limitations is the consideration of pre-mining background levels of PM and total amounts of suspended dust particles during mining. Advancement in the technology leads to the acceptance of instruments with real-time resolution of dust concentration at the mining site. High volume samplers with 8-, 16-, and 24-h monitoring time are used to monitor PM generated during the operational phase; the results thus obtained can be further used to set the limits. Other instruments like dust deposition gauges are also used to estimate the dust load in gm/m² per month during the dust fallout (Pitt et al., 2005).

3.1.2 Water Pollution

During mining, the leftover ore minerals can effortlessly reach nearby water bodies. The pollutants released during the process of mining may contain heavy metals. Heavy metals in drinking water make the water unsuitable for human consumption and drastically affect the region's wildlife. Some mining operations demand the removal of subsurface water reservoirs or aquifers, which can pose catastrophic consequences, such as drying out of water springs, cutting rivers off, and ultimately deteriorating the water quality of lakes, streams, and rivers. For example, the brownish and reddish-orange colour of water adjacent to the mining areas is the first evidence of

the adverse impact of mining on the local water bodies. Subsequently, the pH reduction, increased electric conductivity (EC), high concentration of dissolved ions and heavy metal contamination, reduced dissolved oxygen (DO), and elevated biological oxygen demand (BOD) are some of the important physiochemical and biological signatures that characterize a polluted water body (Singh, 2017).

Sahoo et al. (2013), studied the impact of Acid Mine Drainage (AMD) from mining sites and spoils and found that heavy metal contamination and siltation are the main causes of water quality degradation in the vicinity of the mining area. They also reported that suspended and dissolved particles would dominantly affect the surface runoff and groundwater quality. In particular, suspended particles (sediments) are the main pollutants contributing to the water quality degradation adjacent to the mining areas. Sediment can settle down on the beds of streams and rivers, affecting the population of aquatic flora and fauna. Leachates coming out due to the oxidization of sulfur or sulphidic ores are highly acidic and can readily dissolve heavy metals during the leaching process.

Since mining chokes drainage patterns, mitigation strategies should be proposed to cope with flooding. Mines can overexploit groundwater aquifers near shafts or open pits, drying nearby wells or groundwater bores. Mining activities must also guard against releasing chemically or radiologically contaminated water into the aquifer.

3.1.3 Soil Erosion

Mining operations directly disturb the geosphere. Mining-associated activities drastically impact the land surface, leaving it more vulnerable to erosion. Mining, especially pit mining, accepted as the most common technique among contractors, extracts raw materials after voiding the land surface. The mining procedure requires land blasting, which ultimately strips the surface vegetation, making the surface vulnerable to soil erosion.

The other accepted mining technique is open-cast mining (Fig. 3), which poses the greatest threat to soil resources. Open cast mining is a surface mining technique that follows the underground operation to extract valuable commercial ores and rocks near the surface (Ghose, 2004). Topsoil of the region, an indispensable component considered for land reclamation, especially in the coal mining areas, must be mined out in the initial process of mining for the reclamation of the area (Kundu & Ghose, 1994). This process contains the utmost importance to fulfill the scantiness of the topsoil in the coal field. As a result, it is necessary to preserve the surface soil to protect the primary root medium from contamination and erosion and maintain its productivity. Sendlein et al. (1983) reported that physiochemical characteristics of the top surface soil could be sustained by keeping the systematic storage conditions; afterward, the surface soil can be used in the degraded area.



Fig. 3 Open-cast mining in Uttarakhand.

3.1.4 Noise and Vibration

Mining operations significantly contribute to the generation of noise pollution, which includes exasperating noise from continuous excavation, ventilation fans, and material movement through transportation in the mining sites (Stansfeld & Matheson, 2003), resulting in discomfort or people living in mining areas or their immediate surroundings (Dubiński, 2013). Appropriate planning during the mining process design is required to keep the noise level under the prescribed limits, especially during the evening and night periods; hence, it will not disturb the residents of neighbouring areas. Blast planning must be part of the mining plan to restrain the ground vibration, which could damage building structures in the adjacent areas.

Mining project planners must follow the prescribed standards to limit the vibrational frequency for the region's ecosystem amenities. In India, noise pollution standards are managed by the Central Pollution Control Board (CPCB), which sets the standards for different conditions. In various countries, noise enhancement through atmospheric thermal inversion during the nighttime is also required to be considered an important part of the mining design plan (Lynch et al., 1984). Noise pollution can be an issue because mines are under continuous operation; surface mines generate more noise due to excavation and transport, whereas underground mines produce noise due to ventilation fans, surface facilities, and product transport mechanisms.

3.1.5 Impacts of Mining on Vegetation

Mining has a conspicuous impact on the region's landscape and biological community; the mining operation disrupts vegetation and, consequently, creates a harsh



Fig. 4 Drought condition prevailing due to mining impact on vegetation cover.

environment for its growth. The region's vegetation exhibits a pattern that reflects a wide range of environmental parameters and temporal factors. It also maintains crucial functions in the region by controlling the vital life-cycle elements such as water, carbon, and nitrogen flow, and it is vital to the local and global energy balance. Soil qualities such as fertility, chemistry, and texture are all influenced by vegetation.

In mining regions, vegetation is frequently subjected to severe human interventions, particularly in surface mining and illicit small-scale mining operations, which could cause irreversible land use/land cover variation. Various mining activities directly and indirectly contribute to the region's vegetation loss. The direct removal of vegetation and the hindrance in plant nutrition, transportation photosynthesis, and respiration due to dust accumulation during the mining quarrying process results in the removal of vegetation (Fig. 4).

3.1.6 Impacts of Mining on Humans

The effect of mining on humans is severe. Mining can cause serious accidents such as explosions, fires, and crumbling of mine tunnels, which results in severe physical, social, and economic damages to the mining project and adversely affects the region's living communities (Kitula, 2006; Chauhan, 2010). Particulate matter, chemical effluents, metal contamination, and radiation can affect mine workers severely and can cause lifelong health problems and allergic reactions. Various studies reported that the unhygienic conditions, uncomfortable posture, and heavy lifting resulted in severe injuries to the mine workers, which ultimately reduced the workers' occupational age. Hearing problems and in some cases, deafness also noticed in the mine workers due to loud machinery sounds are also reported (Lawson et al., 2019; Down and

Stocks, 1977). The vibrations from mining heavy machinery and instruments can severely damage the nerve tissue and hamper blood circulation through the brain (Akinunli et al., 2018). Mining of hazardous elements like mercury can cause neurological illness and renal damage among workers (Yard et al., 2012). Exposure to heavy metals like lead, zinc, and copper through air and water pathways severely reduces the lifetime of the mining workers and residents of the neighbouring areas. Inhalation of toxic dust generated in mining processes can result in many respiratory diseases like asthma, bronchitis, pneumonia, asbestosis, etc., (Ikemefuna, 2012).

3.1.7 Impacts of Mining on Biodiversity

Extensive tracts of land and vegetation are removed during mining. The land's viability for farming deteriorates the environment, and animals lose their homes. Biodiversity losses occur in the region due to habitat transformation owing to sudden variations in essential parameters like soil acidity or alkalinity and the local climate (Agboola, 2020).

The area's widespread species are generally affected due to their sensitivity to climatic fluctuation. Mining's impact on biodiversity is a well-known phenomenon that depends upon the bioavailability of different metal ions in the contaminants. When metals in the contaminants are inert, they stay confined in the mining area, but if they are mobile, they can enter the food chain and result in poisoning.

3.1.8 Impacts of Mining on Landscape Degradation

Mining and ore processing alter the terrain in various ways (Farjana et al., 2019). Variations due to extracted materials and dispersion due to mining adversely impact the natural environment arrangement (Fig. 5). Sudden high inflow of certain chemicals generated through mining operations can increase the toxic substances to many folds and hence has a high potential to disrupt the local ecosystem. Unsustainable removal of overburden and waste rock significantly disrupts the region's topography and landscape stability. Numerous studies found that some overburdens may release toxic salts and can also contain oxides of sulfur and phosphorus that can lead to acid mine drainage in the mining region (Musa and Jiya, 2011; Zhao et al., 2022).



Fig. 5 Degraded land due to mining practices.

4 Uttarakhand, A Fragile Himalayan State

4.1 Physiology

Talking about the history of Uttarakhand after independence, this landmass was a significant portion of India's Uttar Pradesh State. However, the inhabitants of this region continued to agitate for a separate state. Consequently, under the Uttar Pradesh Reorganization Act of 2000, a new State separate from Uttar Pradesh was sanctioned (Fig. 6). The areas of Dehradun, Kumaon, Garhwal, and Haridwar have been merged to form a new state of Uttaranchal.

Uttarakhand, formerly known as Uttaranchal, became India's 27th state on November 9, 2000. It is a high-land state in the Central Himalayas, with a diverse climate and vegetation. The state has abundant natural resources, particularly water and forests, numerous glaciers, rivers, deep woods, and snow-capped high summits.

4.2 Biodiversity of Uttarakhand

Uttarakhand has a rich heritage of flora and fauna, which shows the diversity of climate within the state; mountains are covered with snow, highlands with scrubs and semi-high jungles, both dry and wet temperate rain forests and deciduous forests are found in lowlands containing a wide variety of flora and fauna which in turn aids the survival of a diverse range of a species. The state has a distinctive variety of flora; at least 175 species of just plants with healing qualities and aroma are found in the hills and valleys of Uttarakhand, which are very rare worldwide. Possessing

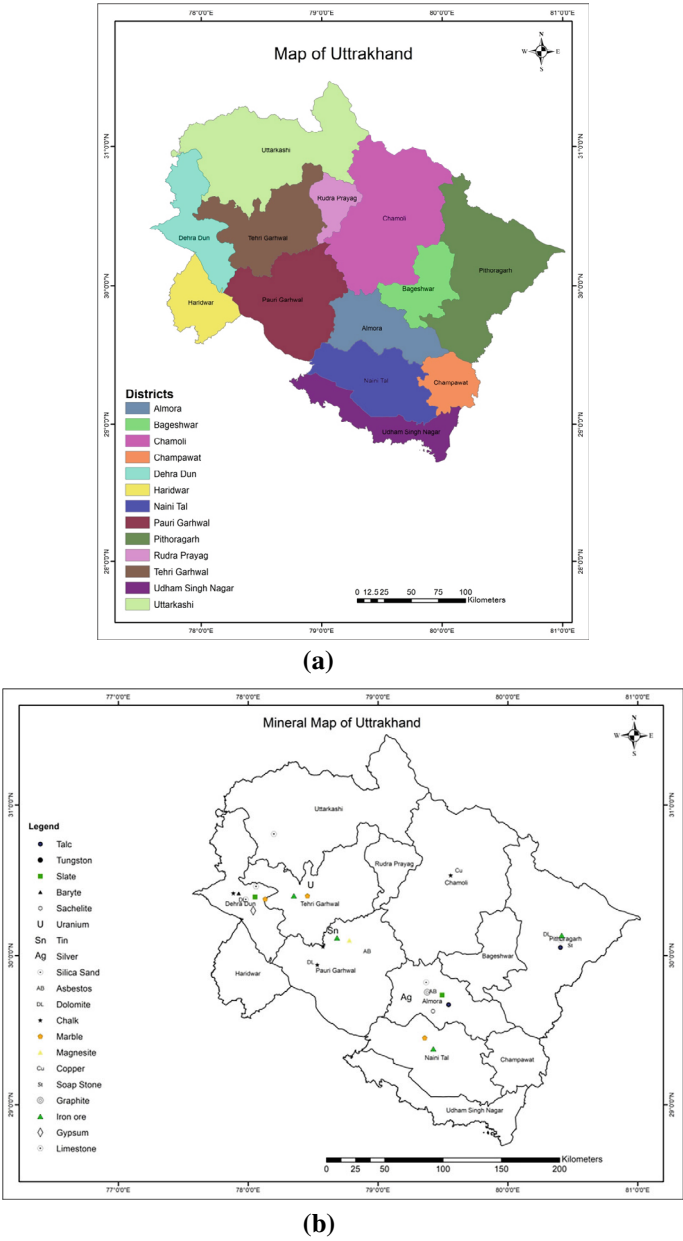


Fig. 6 **a** District Map of Uttarakhand state. **b** Different types of minerals in the state of Uttarakhand

almost all climatic zones, the state has various business opportunities in horticulture, floriculture, and agriculture.

4.3 Minerals Found in Uttarakhand

Uttarakhand is a mineral-rich state. In the state, various minerals are found (Table 1; Lamare and Singh, 2016), both organic and inorganic (Fig. 7). Some of the common minerals are listed below.

- Limestone is found in Dehradun, Masoori, Mandram and Barkot.
- Gypsum is found in Dehradun, Nainital, Pauri, tihri, and Almora.
- Iron ore is found in Nainital, Chamoli, Pithhoragarh, and Tihri.
- Graphite is found in Almora, Pithhoragarh, and Nainital.
- Copper is found in Chamoli, Pithhoragarh, Tihri, Dehradun, Nainital, and Almora.
- Soapstone is found in Chamoli, Bageshwar, and Pithhoragarh.
- Magnesite is found in Uttarakhand's Chamoli, Bageshwar, and Almora districts.
- Marble is found in Masoorie, Dehradoon, Nainitaal, and Tihri.
- Chalk is found in Dehradun, Tihri, Pauri and Chamoli.
- Rock phosphate is found in Dehradun, Tihri, Pauri and Chamoli.
- Dolomite is also known as Portland cement. It is found in Dehradun, Tihri, and Pithhoragarh.
- Asbestos is found in Pauri and Almora.
- Silica sand is found in Pithhorgarh, Almora.
- Gold ore is found across the Sharda, Ramganaga, and Pindoor rivers.
- Silver is found in Almora.
- Tin is found in Chamoli.
- Uranium is found in Tihri.
- Sachelite is reported to occur in the Ranikhet area of Almora district
- Barites are found in Dehradun.
- Slate is found in Almora.
- Tungsten exists in the Chamoli district of Uttarakhand, and 0.13 million tons reserves at Karauli and Alai in the Almora district.
- Talc is a very soft mineral. It is found in Pitthoragarh and Almora.

Soapstone is another mineral found in Uttarakhand's Bageshwar, Chamoli, and Pithhoragarh districts. Among these, Bageshwar district has the richest spot of soapstones. According to reports, "The State Government has sanctioned 90 soapstone mines in this area, majorly found in Kapkot block". However, 75 % of them are operational today. Bageshwar district of Uttarakhand is one of the richest spots of soapstone Digging for soapstone in the vulnerable Himalayas.

Table 1: Different impacts of mining on the Environment

<i>Sr. No.</i>	<i>Type of Mining</i>	<i>Type of ecological Impact</i>	<i>Status of mining</i>	<i>Place</i>	<i>Reference</i>
1.	Limes one	Groundwater Pollution, subsidence, Habitat destruction, Dust	High	Dehradun, Masoori, Mandram and Barkot	(Lamare and Singh, 2016)
2.	Gypsum	Destabilizing slope, Soil erosion	Moderate	Dehradun, Nainital, Pauri, tihri, Almora	(Omoti and Keriko, 2016)
3.	Iron ore	formation of sinkholes, deforestation, destruction of habitat, air pollution, water pollution	Moderate	Nainital, Chamoli, Pithhoragarh and Tihri.	(Khomenko et al., 2017)
4.	Graphite	Sinkhole creation, biodiversity loss, soil pollution, groundwater pollution and surface water pollution.	Low	Almora, Pithhoragarh and Nainital	(Lutz and Buechler, 2020)
5.	Copper	Poisoned soil, affect aquatic biodiversity and farmlands, lungs and heart diseases	Low	Chamoli, Pithhoragarh, Tihri, Dehradun, Nainital, and Almora.	(Liu et al., 2020)
6.	Soap Stone	Soil erosion, deposition, accelerated weather.	High	Chamoli, Bageshwar, and Pithhoragarh	(Khomenko et al., 2017)
7.	Magnesite	soil functions, particularly reduced water penetration rate.	Low	Chamoli, Bageshwar and Almora	(Yang et al., 2012)
8.	Marble	Plants and trees die as marble dust deposition on their leaves causes the extinction of flora and wildlife.	High	Masoorie, Dehradoon, Nainital and Tihri	(Ako et al., 2015 .

(continued)

Table 1: (continued)

<i>Sr. No.</i>	<i>Type of Mining</i>	<i>Type of ecological Impact</i>	<i>Status of mining</i>	<i>Place</i>	<i>Reference</i>
9.	Chalk	erosion of soil, sinkholes, biodiversity loss, or the pollution of soil, groundwater, and surface water	High	Dehradun, Tihri, Pauri and Chamoli	(Baxter, 1985)
10.	Rock phosphate	It violently alters the environment, irreversibly changing the natural landscape.	Low	Dehradun, Tihri, Pauri and Chamoli	(Li et al., 2022)
11.	Dolomite	Landscape degradation, natural drainage damage, pollution, and natural habitat destruction	Low	Dehradun, Tihri, Pauri and Chamoli	(Trivedi and Chakraborty, 2001)
12.	Asbestos	Air pollution, water pollution, soil pollution, Respiratory issues Mesothelioma/ lung cancer Breathlessness Weight loss Vomiting	Moderate	Pauri and Almora	(Goldberg and Luce, 2009)
13.	Silica Sand	Exploitation of land resource	High	Pithhorgarh, Almora	(Orr and Krumenacher, 2015)
14.	Gold Ore	inconceivable amounts of waste, large amounts of Habitat Destruction	Moderate	Sharda River, Ramganaga and Pindoor River	(Ogola et al., 2002)
15.	Silver	soil erosion, deforestation, and water pollution	Moderate	Almora	(Studnicki and Schecter, 2010)
16.	Tin	land loss, soil pollution, and craters filled with acidic water.	Moderate	Chamoli	(Kupka et al., 2020)
17.	Uranium	Water pollution and lung cancer.	Low	Tihri	(Pereira et al., 2014)

(continued)

Table 1: (continued)

<i>Sr. No.</i>	<i>Type of Mining</i>	<i>Type of ecological Impact</i>	<i>Status of mining</i>	<i>Place</i>	<i>Reference</i>
18.	Sachelite	Soil compaction, alkalization, and decreasing soil organic matter, nitrogen, and phosphorus contents.	Low	Ranikhet, Almora district	(Nascimento et al., 2021)
19.	Barites	the exploitation of fertile farmland, very vast and deep trenches are left behind, which might become death traps.	Moderate	Dehradun	(Sikakwe et al., 2021)
20	Slate	Instability in slope, Rock erosion	High	Almora	(Peiffer et al., 1997)
21.	Tungsten	Acidify soil, Lung cancer	Low	Karauli and Alai in Almora district.	(Zheng et al., 2020)
22.	Talc	Water contamination, landscape degradation, lung cancer.	High	Pithhoragarh and Almora	(Banerjee et al., 2022)

4.4 Ecological Impact of Mining in Uttarakhand

The ecological impacts of mining in Uttarakhand have resulted in a loss of landscape and biodiversity and have led to air pollution, dust pollution, water pollution, flash floods, and the wastage of land that is neither suitable for industrial use nor for agriculture. It has also led to many health hazards associated with leftover material from mining.

A study was carried out to investigate the ecological disturbance at two mines, namely Sahastradhara and Chunakhala, located in Doon Valley and Mussoorie hills of Uttarakhand in 1988-1989 (Rajdeep et al., 2011; Lamare and Singh, 2016) and found that mining has adversely affected the ecology, water systems, and floral-faunal population in the area, which eventually affected the human life as well. To deal with these environmental hazards, eco-restoration practices should be implemented to rejuvenate these sites. Eco-restoration is the redevelopment of a degraded ecosystem or restoring the total ecosystem, including all its biotic and abiotic components.

In Uttarakhand, large pits and fissures in the earth's surface were created during sand and gravel mining operations. Mining can sometimes go far deep,

No illegal quarrying, abattoirs to be allowed, asserts Trivendra

TRIBUNE NEWS SERVICE

HARIDWAR MARCH 30

Chief Minister Trivendra Singh Rawat today assured of major clampdown on illegal quarrying and slaughterhouses in the state. Trivendra Singh Rawat said his government would be strict in dealing with illegal quarrying being done on the Ganga riverbed and also take stringent action against illegal slaughterhouses.

"Any one who will work against the law will face action. Our government is strict on corruption and illegal activities. We want to give a good governance and make the state law-abiding one," stated the CM. Meeting Guru Mahamandaleswar Swami Ardeshanand Giri Maharaj at his Kankhal ashram, the Chief Minister held talks on spiritual aspects.

Later, he paid obeisance at Jagadguru Ashram, Kankhal, where he met Jagadguru Shankaracharya Raj Rajeshwarshram.

He also paid tributes to ashram founder Brahmalen Shri Prakashanand Maharaj at his samadhi on the premises. The Chief Minister assured the



Chief Minister Trivendra Singh Rawat pays obeisance to Jagadguru in Kankhal on Thursday. PHOTO: RAMESHWAR GAUR

More facilities for Char Dham

■ The government will be strict in dealing with illegal quarrying being done on the Ganga riverbed and also take stringent action against illegal slaughterhouses

■ Steps will be taken for the renovation of temples and mutts

■ Elaborate arrangements will be done for the Char Dham pilgrimage, along with Hemkunt Sahib

Shankaracharya of taking conservation and renovation of temples and mutts.

On the upcoming Char Dham pilgrimage along with Hemkunt Sahib, Rawat assured them of providing elaborate arrangements for the pilgrimage.

He added that he had given directives to the departments concerned to train the officials and employees deputed or related with the Char Dham pilgrimage so that better coordination and basic amenities were provided for the world famous pilgrimages.

Jagadguru Shankaracharya Raj Rajeshwarshram urged the Chief Minister to cater to all segments of society and overall development of the state.

State to launch app to check illegal sale of river bed materials

Deep Joshi

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DEHRADUN: The state government has decided to launch a mobile application to help mining, police and trade tax department officers track transport of river bed material. The app aims to check the sale of material on the black market.

Mining director Vinay Shankar Pandey said the details of vehicles transporting river bed material have been fed into the app.

"It will be mandatory for a driver to fill a form detailing the nature of the riverbed material, quantity and destination," he said.

"Employees manning electronic weighing machines at the gates of each river will confirm the details," he said.

Pandey said the app will initially be introduced in four districts — Dehradun, Haridwar, Nainital and Udham Singh Nagar. "The app is specific to these four districts because the maximum cases of illegal riverbed mining are reported from there," he said, adding that if successful, the app will be introduced elsewhere as well.

The app will also mention the route to be taken by the driver as well as an estimated time required to transport the materials. This, Pandey said, will help officials notice transgressions in route and travel time.

The state, which plans to introduce this app "soon", will also make it mandatory for the drivers to download and install the app. "It will also be compulsory for all such vehicles to be duly registered with trade tax department," Pandey said.

Fig. 7 Action taken against illegal Mining by Uttarakhand Government to prevent river bed, a local newspaper cutting.

contaminating groundwater, springs, and underground wells. During flash floods, soil erosion, and siltation can be seen (Maliva et al., 2010). Water pollution and blockage of water channels due to mining activities were also reported in many areas due to the dumping of mining waste and overburdening material into adjacent streams. According to the state department of geology and mining, more than 100 sand mines in the state operate along the roadside, causing hill slopes to destabilize (DST, climate 2019). It has adversely affected the Ganga River's ecology and environment in shallow region areas such as Haridwar. Most of the state's plant diversity is badly impacted by anthropogenic disturbances due to many stone crushers and other mining-related problems.

The improper and unsustainable mining activities can cause more damage to the environment by disrupting the equilibrium between the soil and root anchoring. Sharma & Kamboj (2019) investigated the impact of uncontrolled river bed mining on floral diversity and reported a decline in riparian flora composition in the Mohand

Rao mining area. Riverbed mining harms floral diversity. It slows down the succession process in the area. The enormous damage caused by mining to the riverine environment and the adjacent forest had a high ecological impact (Irengbam et al., 2017). The release of mining effluent increases turbidity and nutrient load in the water, which substantially impacts zooplankton distribution and abundance (Kamboj & Kamboj, 2020). A river and a riverbed's physical and chemical properties are altered due to riverbed mining in Uttarakhand. The habitat of zebrafish in Uttarakhand's Nimmi River has been changed due to continuous riverbed mining and habitat disturbance (Sati et al., 2017).

Sand mining activities in the state on a large scale increase the river flow velocity and trigger river bank erosion, which has severe consequences for the river environment (Chopra, 2014). The Wildlife Institute of India (WII) researched the impact of sand mining in the Gaula River in the Nainital area and found detrimental ramifications on the river ecosystem and its biodiversity. Uttarakhand, a state in the central Himalayas, is rich in biodiversity, with a wide range of rare and endangered plants and animals (Kamboj et al., 2012).

4.5 State Policy of Mining in Uttarakhand

The Uttarakhand region is known for its extensive river network and is a mining hotspot. Both legal and illicit mining takes place here, resulting in sand and rock excavation, with an annual revenue of above 2000 crore. People in the state have a long history of fighting illicit mining from the powerful mining mafia, which includes miners, officials, and politicians. When unregulated and illicit sand mining became more prevalent in the state, the Uttarakhand government established its first-ever river training policy in 2016. Mining was prohibited within 100 meters of both sides of the rivers, including downstream and upstream, irrespective of burial grounds. According to laws stated by the Environment Protection Act of 1986, mining silt, sand, gravel, and boulders from the river requires environmental clearance (EC). To combat illegal mining, the Supreme Court, in 2012, ordered that all river mining leases, regardless of the area, must go through the environmental clearance process. In 2016, the Environmental Impact Assessment Notification was amended to include this requirement. If any river mining patch is located in a forest region, the contractor must obtain an extra forest clearance certificate under the Forest Conservation Act of 1980, or FCA (Maliva et al., 2010).

Previously, when Uttarakhand was a part of Uttar Pradesh, unmeasured and unmarked lands and other wastelands were designated protected forests under the Indian Forest Act of 1927. These lands were classified based on the 1893 government notification, reiterated in the year 1997 order, which shows that the FCA applies to all the legal forest areas, even after splitting from the mother state.

Riverbanks are undemarcated lands and are, therefore, considered protected forests. Under the FCA, state governments are banned from issuing any order enabling non-forest usage of forest areas without prior consent from the Union

government, including mining in forest areas. Any activity, such as mining in protected forest lands, would also require forest clearances under the FCA (Maliva et al., 2010).

4.6 Current Status of Mining in Uttarakhand

Unplanned mining operations have polluted the Shiwalik region of the Himalayan glaciers, rivers, streams, lakes, forests, and air, putting their survival at stake, said the high court of Uttarakhand. In response to a Public Interest Litigation (PIL) filed by the residents of Ganwa Village Simroli in the Bhageshwar district in the year 2017 against illegal mining practices, the high court issued a four-month complete ban on mining activities in the forest areas, rivers, rivulets, and streams. The order further prohibited the mining activities in the interstate rivers such as the Ganga, Yamuna, Kali, Dabka, Kosi, and Gaula, along with roughly 100 seasonal rivers and rivulets in the state's foothills (Maliva et al., 2010; Singh et al., 2019).

Again, in 2022, an ashram (hermitage) named Maitri Sadan filed a case in the high court to stop mining and stone crushing along the Ganga in the Haridwar region (a pious place for Hindus). As a result, the high court ordered the state government to temporarily halt all mining operations between the Raiwala and Bhogpur areas of the district.

The state modified its River policy in 2020, making significant changes in the environmental laws.

4.7 Mitigation Strategies

It's now an accepted fact that mining and related activities have had some undesirable impact on the environment. Efforts should be made to make the next generation of mining practices more advanced and eco-friendly than the ongoing ones. The following procedures can be implemented to significantly reduce emissions during mining operations.

The mining sector is highly energy-demanding. If feasible, it should be replaced with renewable energy sources. A more efficient leaching process using minimal inorganic chemicals should be used, thus reducing the impact on human health, both carcinogenic and non-carcinogenic (Merem et al., 2017; Overgaauw et al., 2020).

It's also necessary to stabilize, replant, and revitalize mined areas with suitable plant species, thus restoring the green cover. The toxic discharge should not be allowed to seep into the ground and contaminate the groundwater. Proper treatment should be done with the latest technologies and processes before releasing it back into the ecosystem.

The latest use of information technology can be very helpful in reporting and tracking the illegal mining activities in the area by the local residents. The Uttarakhand government has launched a mobile application to monitor such activities. To monitor unlawful mining, several geo-surveying technologies that employ drones or satellite data might be utilized.

In hilly areas, 'strip mining' is more prevalent, destroying the hills' landscape and beauty. To combat this, a practical engineering method to minimize the landscape degradation caused by mining is known as the tunneling Method. In this technique, ore deposits can be excavated by constructing tunnels at the bottom of the hills and combining them with cross cuts and adits. This technique protects the vegetation cover and also reduces dust produced during mining activities (Dwivedi & Soni, 2021).

5 Conclusions

Natural resources are a long-lasting treasure for human well-being, but ever-increasing demand for a luxurious life causes these resources to be overexploited, resulting in environmental deterioration. Due to environmental concerns, the government has banned many mining sites in the state, but still illegal mining is prospering in hard-to-reach pockets. Although minerals extracted from mining operations have a significant influence on the state's economic development, at the same time, mitigation measures should also be implemented in order to minimize the negative impact on the environment. The state government has employed various initiatives, such as preparing environmental management plans before the implementation of mining projects and launching a mobile app named "Mining Guard" to track all legal and illegal mining activities. As per the Department of Mining and Geology reports, the state government lacks monitoring, accountability, and transparency in mining operations, which directly enhances illegal mining in the state.

To combat the environmental hazards associated with various mining activities, such as air pollution, water pollution, and landscape degradation, government agencies should not grant mining permission without first obtaining environmental clearance, and environmental policies should be revised over time so that the environmental effects of mining activities can be reduced to the minimum. A committee of highly scientific people must be created to monitor the enforcement of laws related to environmental protection in an efficient and unbiased way.

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Chapter 7

Risk Exposure to Mercury (Hg): Factors Associated in Sentinel Animal Models Exposed to Small Scale Artisanal Mining Areas (ASM)



Avila-Rodriguez Albert and Luna-Acosta Andrea

1 Introduction

The release of mercury (Hg) in small-scale artisanal mining areas (ASM) continues to be the primary source of anthropogenic emission of that element globally (UNEP, 2019). Hg has a series of physicochemical properties that allow it to go through various stages of speciation, both organic and inorganic (Selin, 2009), increasing or decreasing its toxicity on exposed organisms (Zillioux et al., 1993). Emissions from tailings and mining operation areas are estimated to exceed 1,000 tonnes of Hg annually (Esdaile and Chalker, 2018). Despite the above, this activity continues to be a means of subsistence for millions of people in several countries (Cossa et al., 2021), which is why it is necessary to develop monitoring methods that allow the generation of action plans to reduce their impacts. Under these antecedents, Hg continues to be linked to the development of different diseases worldwide (Chen et al., 2016; Laks, 2009; Lieske et al., 2011), being considered a risk factor for their occurrence, mainly in mining sites related to the ASM. Regular exposure to mercury vapour during amalgam handling is critical for developing complications associated with this element, which could occur under different scenarios. In the first place, the emissions that are produced in the gold shops where the amalgam is commercialised for semi-industrialized processing. Second, it is carried out in urban homes by hand, affecting many people directly or indirectly linked to this activity (Cordy et al., 2011; 2013; Esdaile and Chalker, 2018).

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For Latin American countries impacted by illegal mining, such as Colombia, gold extraction using amalgamation methods is the primary source of Hg emissions (Olivero et al., 2004; Olivero and Jhonson, 2002), resulting in being seen as a growing problem due to the high concentrations that have been documented in humans (Marrugo-Negrete et al., 2008). Stopping mercury contamination in ASM is often a complex process. So much so that restrictions on the use of this metal do not seem to be an effective way to solve this problem due to the documented evidence of the limited response obtained in countries where the legal prohibition of its use has already been implemented (Hilson, 2006; Sousa et al., 2011), this situation could have several explanations, such as the high value of this mineral in the market, the illegality of the extraction processes, the limited financial resources for research, and the lack of participation and direct support for the miners by government entities, among other causes. Under these precedents, the scientific community sees the development of methodologies that allow advances in control, surveillance, and mitigation strategies. The foregoing becomes essential if this research introduces low-cost components with high technological development that are robust, scalable, *in situ*, and particularly novel (Esdaile and Chalker, 2018).

Under the growing evidence of high Hg emissions by ASM on the planet (UNEP, 2019; UPME, 2020), it is necessary to determine the study of new routes of exposure to mercury as well as its consequences and effects on humans, for which the development of early warnings of exposure is required to facilitate decision-making on the risk factors to which human beings in a population near areas of emission of environmental pollutants could be exposed (NRC, 1991). Although it is believed that diet remains the leading risk factor for the generation of adverse effects of Hg, other vital routes of exposure that have been well documented, such as the atmospheric route, cannot be ignored (Cordy et al., 2013), which represents a source of low concentration of this metal. However, over time, it could trigger consequences similar to those reported for oral exposure, especially in terrestrial animals which are frequently exposed to this route, mainly in areas close to mining zones (Lyatuu, 2002; Kinabo and Lyatuu, 2009; Toyomaki et al., 2020), which is why the development of monitoring tools would become a useful tool to assess these risks. Exposure to environmental chemical contaminants in domestic animals, especially canines, has been documented over the past 90 years. In particular, the use of animals as tools to assess exposure to different pollutants such as pesticides, brominated compounds, polychlorinated biphenyls, and heavy metals such as mercury and lead has been suggested (Avila et al., 2023).

In this context, the use of sentinel species in activities related to mining dates to the mid-1980s, with the canary being one of the first documented species, not changing until the end of the same decade when technological development allowed the use of detectors automatic (Beck et al., 2020). Officially, in 1991, the National Research Council detailed the use of animals as sentinel species, considering animal surveillance as an additional tool for evaluating public health risks. Being feasible to use both to monitor the impact of toxic exposures of a different nature in humans (Schmidt, 2009; NRC, 1991), such as to provide information on changes in the

incidence of diseases, their spread, and risk factors on a specific population (Schmidt, 2009).

Today, the role of animals as sentinels of human exposure to environmental toxins has been well documented (Pastorinho and Sousa, 2021). Thus, the relationships between exposure and tissue concentrations can be considered for applying predictive models (López-Alonso, 2018), thus considering wild and domestic species, both for production and companionship, ideal for evaluating the response to long-term toxins in different scenarios (Abbott, 2017).

In this context, the main objective of this chapter is to analyse what criteria should be considered when choosing the most appropriate sentinel animal model that shares similarities with humans, important biomarkers, exposure pathways, and adverse effects in ASM areas. The above are early warning and control systems that generate short-term results that can be used as alternative monitoring and surveillance strategies for global mining problems.

2 Principles of Animal Sentinel Modelling

2.1 Contextualization of Hg Contamination in Mining

Mining continues to be a profitable activity for the economic development of many countries; however, it is poorly planned execution, coupled with the role of illegality in some regions of the planet, leaves with it a series of adverse effects that today have managed to be well documented (Padró et al., 2019; Lara-Rodríguez et al., 2020). As will be stated later, the low number of investigations, added to the scarce reporting of data related to the monitoring and surveillance of mining activities, have allowed the generation of great uncertainty about the number of people involved and the amounts of environmental contaminants that could be issued. Taking the above as an example, for the first scenario, it is estimated that between 40 and 150 million people (Eftimie et al., 2012) in more than 100 countries around the world practice activities related to ASM, concentrating their activities on low- and middle-income countries (Artisanal mining, 2020). For the second scenario, it is estimated that by 2050 the emissions of this metal in the world would exceed 2,390 to 4,860 Mg/year (Streets et al., 2009), of which it is expected that most of these will come from related activities. With ASM (UNEP, 2019), under this framework, there are still information gaps based on health and safety measures in work processes, including ASM. The development of more robust health monitoring studies allows the retrospective and prospective development of information regarding the dynamics of pollutants such as Hg and their deleterious effects on people's health.

Today, exposure to Hg is linked as a risk factor in the genesis of different pathologies, mainly in communities surrounding ASM (Rose, 1944; Cole, 1944). However, a relationship between urban and suburban communities has also been seen due to the socio-economic dynamics interconnecting them and this element's volatile and

unpredictable behaviour (Cossa et al., 2021). Environmental contamination related to ASM is strongly associated with specific regions of the planet, such as Canada, the USA, Africa, China, the Philippines, and South America (Eisler, 2004), for which the development of studies linked to sentinel animals would have broad applicability on different continents due to the wide geographic distribution of some species, particularly domestic ones, primarily due to the historical relationship they share with humans (Taborda, 2020; Sanders, 2003).

2.1.1 Toxic Metals in Animals

Concentrations of Hg have been found in different organic and inorganic matrices such as sediments, plants, and animals, in many cases exceeding the limits allowed by health entities worldwide (Eisler, 2004; WHO, 2017). In the case of animal organisms, the appearance of symptoms derived from acute exposure to Hg will depend on several factors, such as the route of exposure, the sensitivity of the species, the health status of the exposed organism, and the type of speciation in which the xenobiotic is found in the environment (organic and inorganic Hg), among other components that explain it (GreenWood et al., 1990; Blakley, 2021). It is estimated that exposure to this metal may require months or years to trigger bioaccumulation processes in the body (Beck et al., 2020). However, the dynamics of this element in the face of chronic exposures and different and combined exposure routes are unknown. Therefore, more research is required on animal species that allows us to get closer to fulfilling the above assumptions in the short term (Abbott, 2017).

In general, exposure to Hg is linked to the appearance of haematological, gastrointestinal, hepatorenal, and neurological toxidrome (Camerino et al., 2002; Wedeen, 1989). However, there are limited reports that inform health authorities about the associations between the health of domestic or wild animals, as biological monitors of exposure to this metal (Beck et al., 2020), thus ignoring the relationship between environmental contaminants such as heavy metals (Abbott, 2017; Pastorinho and Sousa, 2020) and biomonitoring in small-scale artisanal mining areas. For this reason, a necessary call is made to the different medical and environmental disciplines to understand the research importance of animals as early warning systems of risk factors that may be applicable in humans when exposed to heavy metals, such as Hg, due to a series of advantages that will be described later in this document.

2.2 Sentinel Modeling

The use of sentinel species information has been provided by several organisations worldwide, such as the US Exercise Environmental Health Research Center, the EPA's National Center for Environmental Assessment, and the US Environmental Protection Agency for Toxic Substances and Registry Diseases, being considered as a feasible model to determine the potential effects of chemical products in the

environment, ideally to evaluate causes and effects. However, this type of mechanism cannot be considered the only tool for evaluating the effects of contaminants. On the contrary, it should be seen as an additional tool for developing risk assessment studies and monitoring the effectiveness of remediation activities. On the contrary, it should be perceived as a supplementary tool for advancing risk assessment studies and monitoring the efficacy of remediation endeavours. Despite the gradual acceptance within the scientific community, the outcomes documented in various studies have provided momentum for enhancing the model over time (Van der Schalie et al., 1999), now acknowledged as an instrument for promoting public health (Pastorinho and Sousa, 2020).

By 1991, and as a starting point, the Environmental Studies and Toxicology Board of the National Research Council (NRC), together with the Commission on Life Sciences, made a thorough review of the potential use of sentinel species to evaluate the environmental health, considering these data useful for making regulatory decisions. Together, the development of other tools is required to strengthen and optimise the sentinel model, such as using specific biomarkers that respond to and guide biological events that would probably occur in humans (Van der Schalie et al., 1999), improving the results of the investigations. Despite this, some considerations need to be thoroughly inspected before choosing the most appropriate sentinel species to develop modelling studies, such as tolerance phenomena or sensitivity to toxins that species present (NRC, 1991). For example, the death of a fish by residual chlorine would have few or no significant effects on the human species. Likewise, the presence of exposure phases that would be unlikely for humans should be considered, such as the exaggerated consumption of sediments contaminated with non-bioaccumulative carcinogenic substances in a catfish. This situation would be unlikely as a potential risk for humans (Van der Schalie et al., 1999).

To carry out rigorous animal sentinel modelling, it is required that the contact with the toxin be like that described in humans (Beck et al., 2020), with the use of higher animals, domestic and wild, the most frequently used to comply with this assumption (Abbott, 2017). In this sense, and despite the above, invertebrate animals are also helpful in investigating and understanding contaminant behaviour and enzymatic pathways of toxins due to the ease of sampling and obtaining analyses under laboratory conditions. For example, the use of oysters and other bivalves allowed the identification and detection of metalloinducible proteins, such as metallothioneins, due to the affinity of the accumulation processes of these metals (Langston et al., 1998), being useful biomarkers for the evaluation of monitoring and remediation activities in specific areas. Attempts to monitor the impact of environmental contaminants such as Hg can focus on using the same human species and other species. However, there are still information gaps that require the use of new tools different from those studies that have been developed on human beings, which allow for providing more information under other types of scales and perspectives (O'Brien et al., 1993; NRC, 1991).

2.2.1 Advantages of the Sentinel Model

Using sentinel species as indicators of exposure to environmental contaminants has several advantages such as the direct collection of different types of matrices, both invasive and non-invasive (hair, blood, plasma, serum, etc.), the bioavailability and detection of the xenobiotic on the exposed animal, and the ease of acquiring samples in routine procedures, among some others (NRC, 1991), especially in companion species. The above is due to several qualities, such as a short and compressed life expectancy, complex tissue sampling that can be performed at the time of death of the animal or during routine veterinary procedures, with the consent of the owner, and professional guidance in animal health (NRC, 1991). This situation becomes relevant due to the high degree of medical development that some domestic species present, surpassed only by humans (AVMA, 2012), substantially improving diagnostic processes.

Thus, the affinity of some animals to the exposure of a series of toxins commonly found in the environment and that their exposure would have severe repercussions for human beings is described, but not without first generating early warning systems already evidenced and documented. (NRC, 1991; Backer et al., 2001; Beck et al., 2020). Some examples of the above are the cases of intoxication in birds due to inhalation of Teflon polymers (Blandford et al., 1975), cats due to exposure to Hg from waters contaminated by this metal in Minamata Bay, and the use of canine sentinel model for exposure to lead and aromatic hydrocarbons, among others (Kurtz and Kim, 1972; Pastorinho and Sousa, 2021; Toyomaki et al., 2020).

Against this background, it should be made clear that not all toxins to which animals are exposed result in symptoms of disease or vice versa, taking as an example the presence of some domestic plants that are toxic to animals but are harmless to humans (Beck et al., 2020). Likewise, the susceptibility of some animals to presenting symptoms should be considered due to the intrinsic nature that they possess when ingesting food indiscriminately (Swirsky et al., 2020) or due to the inefficient metabolic processes of the species to process the xenobiotic (Allen, 2003). Under this premise, and as seen below, sentinel animal modelling continues to be a tool that should be well-discussed to improve data reliability that can be extrapolated and applicable to humans. Similarly, toxic surveillance using animal poisoning reports has come to be considered an essential resource for obtaining real-time poisoning data, thus providing information on the presence and trends of emerging toxins in the US, information that could be extrapolated to the sentinel animal model if similar reports were achieved under conditions and data on environmental contaminants in specific regions (Swirsky et al., 2020).

Despite the above, Martín et al. (2015) and Schramm (2008) show a certain mistrust of the use of non-invasive samples for the study of environmental contaminants due to the differences between the quantities absorbed by the organisms exposed to different variables such as the atmospheric environment, ingestion, or absorption behavioural dynamics of toxins in the environment, as well as their levels on other body fluids (González et al., 2018). Although toxicity studies under laboratory conditions are helpful in determining dose-response interactions, they would not provide

information on the integration of some components, such as chronic diseases, stressors, and variable exposure to real concentrations of mixtures of environmental chemicals, which commonly precede the appearance of irreversible clinical responses, disease processes or the death of exposed organisms (Fox, 2001).

Despite the foregoing, it's essential to acknowledge the inherent value of animals as sentient beings. Consequently, the conduct of investigative studies necessitates regular oversight by bioethics institutions. These bodies ensure that such endeavours are conducted in a manner that minimizes stress, pain, illness, hunger, and thirst among the subjects, while also facilitating the expression of natural behaviours within their environment. These objectives align with the principles articulated in Brambell's five freedoms (1965) (FAWC, 1993).

2.2.2 Potential Sentinel Modelling Mammals

The environmental implications that mining exerts on the environment seem incalculable (Padró et al., 2019). It is estimated that the damage to predators, birds, and mammals seems to be relevant. This is because many of the mining sites are close to biodiverse regions such as the Amazon rainforests (Esdaile and Chalker, 2018), which is why the use of wild animals has managed to be considered as classic sentinels of environmental health risks as they share the same subsistence resources with humans. Birds, fish, amphibians, and turtles are among the wild species most implicated in this type of study (NRC, 1991; Schmidt, 2009). However, as mentioned above, the choice of the appropriate sentinel model will depend on both the study's target species and the pollutant's behaviour within the environment.

Production species rarely function as sentinel species for contaminant diseases in humans; instead, they have become valuable in determining zoonotic diseases of public health importance. However, the study of domestic animals for productive purposes has turned out to be beneficial because they are essential sources of food for humans, being considered a direct source of exposure and transfer of contaminants through the ingestion of meat products (Miranda et al., 2003; Nkosi et al., 2021). In this sense, the use of bovines, especially calves, have been considered as biomonitors of soil pollutants such as arsenic, copper, and zinc because they tend to concentrate adequately in sites such as the liver and kidney, being helpful in monitoring these elements in areas where significant concentrations of these elements are presumed to be generated and emitted, making it feasible that this species can be used as a monitor in sites adjacent to ASM areas.

Under these precedents, domestic companion animals (Pastorinho and Sousa, 2020; Taborda, 2020) can become much more effective sentinels for the evaluation of a wide variety of toxins since they share a common environment with their owners, being considered as potential in the early identification of contaminants, infectious diseases, chemical terrorism events among other faculties (Schmidt, 2009). Despite this, the most significant limitation for the use of sentinel animal species continues to be ethical, legal, and sometimes logistically impossible issues, and there is also little information related to the condition of ecosystems and public health, for which

greater understanding is required in this topic to understand the dynamics of these species (Schwacke et al., 2013), mainly in mining areas where these processes have not yet been efficiently documented (Toyomaki et al., 2020).

2.3 Sentinels of Exposure in Mining

Small-scale artisanal mining remains an important livelihood for supplying economic resources in some regions of the world, mainly for those in the development process (Padró et al., 2019). An analysis of gravimetric material flow allowed us to determine that between 70 to 80% of the Hg used during gold processing is lost in the atmosphere, its highest point of concentration, with lateral or vertical dispersion, as well as soils and sediments, being its main sites of accumulation in the environment. Similarly, tailings, goldsmith centres, and workshops can be considered hot spots for Hg emissions (Van Straaten, 2000), areas in which monitoring, evaluation, and control strategies are required for these activities (Cordy et al., 2013), being sentinel modelling a tool that could remedy these assumptions.

A sentinel species must be large enough to obtain samples that allow the quantification of the toxicant. Likewise, it must have a compressed shelf life, have a diet like that consumed by humans (especially omnivorous species), and be exposed to the same phases of exposure. In the same way, its abundance over the territory should be frequent to carry out and facilitate the corresponding statistical analyses (NRC, 1991). On the other hand, it must have enzymatic processes similar to human beings, such as genetically homologous metabolic pathways and comparable physiological and physicochemical processes. The above is to improve the degree of extra polarity of the results with the target species, in this case, humans (Backer et al., 2001; Beck et al., 2020; Ostrander, 2012; Nkosi et al., 2021). Finally, yet importantly, you will need to maintain your population over time to enhance your reproducibility, as well as be able to live with captive populations. For this reason, using large mammals may not be feasible due to the maintenance costs and the logistical difficulties of maintaining and reproducing them (O'Brien et al., 1993).

It is commonly considered that the sentinel animal species must have a measurable clinical or immunological response, which must be recognisable by the investigator, even though these may not be identical to the clinical signs of disease in humans (Schmidt, 2009; Pastorinho and Sousa, 2020). Sentinel models can be simple or complex systems intentionally placed in areas to extract helpful information through advanced diagnostics, whether through blood tests, serum biomarkers, or genetic studies. Generally, the people who live around the mining titles have animals that can accumulate large amounts of environmental contaminants such as mercury, cadmium, and lead, mainly in organs such as the liver, kidney, and muscle tissue (Chibunda and Janssen, 2009; Taborda et al., 2020). About this aspect and serving as an example, Miranda et al. (2003) managed to find Hg concentrations in organs such as the liver and kidney, mainly in domestic cattle, highlighting for their research the importance

of monitoring programs to determine traces of heavy metals in meat for human consumption in places close to emission sources (Nkosi et al., 2021).

Similarly, Chibunda and Janssen (2009) in Geita-Tanzania, one of the largest districts with mining centres in that country, reported the presence of 436 to 820 ug /kg Hg in liver samples from cattle and birds near these mining settlements, respectively, thus generating a first approximation on the possible implications that reporting these data would have on circumscribed populations (Alonso et al., 2002). In this same sense, Lyatuu (2002) reports the detection of Hg in domestic ducks raised in a mining town of Mugusu-Uganda, which fed near abandoned amalgamation and tailings ponds. Similarly, Kinabo and Lyatuu (2009) came to consider domestic ducks as potentially unsafe for human consumption since it is an omnipresent species that is commonly found in gold mining camps, reporting high concentrations in tissues such as the lung, liver, gizzard, and feathers, which were related to variables such as weight and age of the birds (Swirski et al., 2014). Thus, the results of this same study made it possible to determine that the atmospheric pathway could be another alternative pathway for the diffusion of gaseous Hg due to the high concentrations found in the lungs of this species, recommending continuing to monitor these organs prospectively to relate similar data in humans (Parker and Ostrander, 2005).

In this context, it seems probable that the tailings are considered sources of contamination for Hg, and it would be worthwhile to study sentinel species that allow the dynamics of Hg on these territories to be determined. Similarly, the authors of this research recommend avoiding possessing livestock or animals for consumption near the artisanal mining areas. However, on the contrary, its use is directed as a simple monitoring method to take measures that reduce the risk of contamination by Hg for the populations that circumscribe a mining title (Alonso et al., 2002; Chibunda and Janssen, 2009). Several analyses of Hg in countries such as Spain, Sweden, and Poland found high traces of this metal in organs such as the liver. However, according to daily consumption values for this organ, the values found were considered low risk for humans. According to the documented evidence reported by Chibunda and Janssen (2009), the use of Hg for artisanal gold extraction is discouraged, partly thanks to the results of this study, due to the possible risk they would have on humans and animals that live in these territories.

In this same sense and for other sentinel species, Marienfeld (1979) proposed using domestic pigs near mining areas to evaluate birth defects associated with heavy metals for 3 years, finding a direct relationship between the high concentrations of lead at these sites. However, the existing relationships with Hg and the adverse effects triggered in artisanal mining areas seem to be limited and rare for other domestic species, particularly pets. In contrast, Takeuchi et al. (1977) proposed using domestic cats as potential sentinel specimens for the evaluation of the effects of mercury poisoning in Canada based on the development of neurological signs that these animals presented when ingesting fish contaminated with Hg from river systems in Ontario. Under this scenario, a greater number of studies are required about ASM, particularly in areas of little research, which until now have not been given importance, such as the use of unusual environmental matrices, the use of little-studied sentinel species, and high research potential, such as the use of terrestrial mammals, among others.

Domestic mammals have commonly served as sentinels for disease outbreaks and environmental contaminants of public health concern (Abbott, 2017). Despite the above, we cannot ignore the use of fish as a species that has been used frequently for the development of studies to determine toxic effects in humans. This is because they have many of the expressed genes that are of clinical interest in humans. Studies of toxicity to pollutants and the prevalence of their effects (carcinogenic or genotoxic) have been widely documented for this species, especially the use of zebrafish (*Danio rerio*) (Schwacke et al., 2013). However, the development of studies related to Hg about its neurotoxic effects in fish seems less evident compared to those that can be evidenced in-depth in mammals (Nogueira et al., 2010).

2.4 Biomarkers of Interest for the Detection of Hg Effects

Biomarkers are defined as the biological responses of organisms to the presence or exposure of xenobiotics (Van der Oost and Vermeulen, 2003). Many biomarkers have been helpful in monitoring areas where high concentrations of pollutants are frequently emitted due to their sensitivity to assessing their effects. Metallothioneins, a sensitive biomarker to identify heavy metal exposure, are low molecular weight inducible protein families rich in cysteine-type amino acids that allow regulation of homeostasis in the face of toxic metal exposure (Van der Oost and Vermeulen, 2003). Metallothioneins are considered a marker of importance in research due to the close inherent relationship they have had with different metals of importance in environmental toxicology. Organs such as the liver and kidney have a higher gene transcription rate for metallothioneins due to their fundamental role in the metabolism and excretion of toxic substances. As an example of the above, Zalups and Koropatnick (2000) were able to identify higher gene expression rates in rats treated with inorganic mercury, showing a direct correlation between mercury content and gene expression of this marker.

The use of different matrices and biomarkers have been used as indicators of exposure to mercury in humans, such as the use of cerebrospinal fluid, genetic material, whole blood, hair, nails, and urine, among others, showing a clear relationship between the proximity to the emission sources concerning the concentrations obtained for these matrices (Marrugo-Negrete et al., 2008; Olivero et al., 2004; Cordy et al., 2013). Findings are exhibited as a valuable tool to link exposure factors and the effect of these pollutants on humans, mainly when associated with activities related to ASM (Cossa et al., 2021). It seems that the best way to acquire information on the adverse effects of mercury exposure in humans would not be using the same species due to a series of factors that explain it. In the first place, there are ethical components that would limit its use, such as the use of complex matrices that are difficult to access, such as nervous, reproductive, and endocrine tissue, among others. Second, we found that the latency time to bioaccumulate or express the adverse effects of this metal would require longer time scales. Thirdly, we find the appearance of essential variables, such as changes in behaviour, the presentation of diseases

and clinical syndromes that are difficult to predict, among others, which is why the results obtained will have a series of biases that would modify the results of said investigations (NRC, 1991; O'Brien et al., 1993; Schramm, 2008).

On the other hand, and in a thorough manner, the development of sentinel animal studies must consider both the matrix and the species with which the modelling of the data will be worked for the extrapolation of cause and effect in the human species. For example, invasive matrices such as the liver and kidney should be considered, as well as organs where this type of metal tends to accumulate more, and the above when it is ethically and logistically possible. On the other hand, the use of matrices such as serum would not show the type of total body load of environmental contaminants, as organs such as those mentioned above would. However, their presence would determine the bioavailability and distribution of the xenobiotic in the organism. In this sense, it is necessary to evaluate the reliability and specificity of using biomarkers in animals to predict risks in humans (Backer et al., 2001). Under these antecedents, the use of mammals seems ideal for developing sentinel modeling for the human species due to the metabolic and physiological similarities they share (O'Brien, 1993). However, the choice of the sentinel species will depend specifically on how the contaminant is concentrated in the environment and the availability of the matrix, in addition to the ease of finding the species and its easy handling. In this sense, the use of migratory mammals has been suggested to monitor the environmental contamination of large areas (Kurtz and Kim, 1972). However, it must be considered that the migrations could be limited and will not adjust equitably to the patterns that would be observed with the use of another type of sentinel animal (O'Brien et al., 1993).

Under this framework, it seems essential to rethink the type of contaminant that you want the selected sentinel species to evaluate to assess the effects that would be present in the human species, as well as to estimate its advantages, disadvantages, and limitations. In this way, using domestic animals such as a dog would be feasible due to the similarity between the morphological changes of tissues, age, and the appearance of haematological findings that occur in the human species (Zook, 1973; Thomas et al., 1976). For the case of canines in sites close to Superfund areas, a series of essential biomarkers were selected and related to clinical parameters frequently used for humans. For example, a higher frequency of appearance of lymphocyte micronuclei, CD4 and CD8 lymphocyte typing, as well as alterations in the most used serum biomarkers for clinical use in humans (such as albumin, phosphatases, hepatic transferases, ions, amylases, etc.). Thus reducing the uncertainty associated with predicting contamination risks in humans (Backer et al., 2001) and obtaining much more reliable data that can be considered. Despite this, it is necessary to delve deeper into the appropriate sentinel species to develop sentinel studies in ASM areas.

Specifically, it has been possible to propose the development of models that simultaneously include the use of more than one species in the field and thereby eliminate the differences and limitations usually defined for each species (O'Brien et al., 1993). In the case of ASM, it would be interesting to evaluate species that have the same sources of exposure as humans and animals that concentrate these pollutants on easily obtained matrices such as hair, among other considerations. In this sense,

dogs and cats are suggested due to the historical advantages each species presents about Hg, its high frequency and geographical distribution, and the metabolic similarities with human beings—reasons why they could be inferred as potential species for the development of this type of research.

2.5 Companion and Wild Animals as Indicators of Public Health Toxicity

The use of animals as indicators of toxicity is not a recent phenomenon. There is evidence that animals, mainly companion animals, are exposed to exposure phases and toxidromes like those reported in humans, serving as early warning systems for exposure to toxic substances that are frequently found in the environment (NCR, 1991; Beck et al., 2020; Pastorinho and Sousa, 2021). Consequently, it seems feasible that the evaluation of contaminants in both rural and urban areas can be achieved by monitoring companion animals, which is previously necessary if we highlight exposure to Hg as a risk factor in the genesis of diseases for both humans and animals. Under this scenario, it can be inferred that the use of companion animals as sentinel models turn out to be beneficial, being especially useful to direct study plans in the detection of diseases that are related to their toxicity in the long-term and the short-term periods, thus allowing greater ease for the identification of its adverse effects (Swirsky et al., 2020).

In this sense, in other canids such as red foxes (*Vulpes vulpes*), high concentrations of this element have been found, which is why they have been considered a bioindicator species of Hg contamination for terrestrial ecosystems in northeastern Poland (Kalisinska et al., 2012). Despite the above, few sentinel animal species have been considered or valued for ASM areas, for which it is necessary to carry out research related to this topic to close gaps in knowledge about the dynamics of environmental pollutants such as Hg in urban and rural areas and under other scenarios.

3 Future Directions

Exposure to sources of mercury other than those commonly known as methylmercury should be studied mainly in areas where it is emitted anthropogenically. It is documented that mercury vapour and the air exposure route constitute a representative source of risk for human beings, as it is mainly associated with the injury of essential organs such as the liver and kidney (Nordberg and Nordberg, 2009; Brambila et al., 2002). Several compensatory mechanisms have been identified in the face of exposure to heavy metals such as Hg, nowadays important biomarkers for contaminant monitoring and surveillance strategies in different species, mainly mammals (Nordberg and Nordberg, 2009), such as the use of genes of expression

of metallothioneins and glutathione transferase, essential molecules in heavy metal homeostasis (Brambila et al., 2002). The number of techniques used in experimental toxicology still presents a series of gaps that require their approach by different disciplines. The foregoing becomes more important if we take as an example the lack of research based on the various forms in which metallothioneins are expressed in different species (Nordberg and Nordberg, 2009), as well as their ability to be used as specific biomarkers of research interest (Alonso et al., 2002).

Research in ASM settings should address health transversally and comprehensively, including the possibility of reviewing various aspects, such as the spread of infectious diseases, their effects on mental health (Cossa et al., 2021), including the use of other species that are implicitly and historically linked to human beings, such as companion or production animals, both in urban and suburban communities (Cossa et al., 2021). Although the use of both portable X-ray absorption and fluorescence spectrometry equipment have been used to monitor emission zones linked to ASM, these methods seem to be expensive and not very accessible, which is why the advancement of other technologies, as the use of sentinel modelling would allow for low costs to obtain both prospective and retrospective data in the face of exposure to environmental contaminants.

The high solubility of inorganic mercury species such as mercury cyanide has allowed greater conduction of methylation towards organic species of this element within the food chain, including complex water systems. In this sense, the sum of exposure sources both in humans and in sentinel models would achieve a definitive role in the predictive data that this element could have in different space-time scales (Esdaile and Chalker, 2018), being the development of sentinel modelling a vital tool for the evaluation of results in remediation processes.

The importance of measuring biomarkers lies in the fact that many environmental pollutants tend to affect key organs for the regulation of neurological, reproductive, and metabolic processes, among others (González et al., 2018; Miranda et al., 2003; López-Alonso, 2018), which is why the clinical measurement of their status would allow us to give a general picture of the health status of the exposed organisms, thus allowing the observation of early subclinical responses shortly after exposure to dangerous agents.

Against this background, improving the communication style between the scientific community and public health regarding this research topic is necessary, allowing continuous updating and reliable databases that facilitate decision-making (Schwacke et al., 2013), especially in areas of high socioeconomic and environmental vulnerability. Finally, it is emphasised that the development of this methodology must go hand in hand with alternative support and employment opportunities for miners to be considered as applicable measures that allow for the speeding up of monitoring processes and surveillance of ASM areas and the relevant mitigation measures.

4 Conclusions

The sentinel animal model is an additional tool for mitigating, monitoring, and controlling the emission of pollutants on specific territories. Although using animals as sentinels of toxicological processes in humans in the face of environmental pollutants is not a recent process, its progress in recent decades has allowed the replication of this methodology in some areas of the planet where the emission of pollutants is usually suspected. However, variables such as the choice of the type of matrix and the discrimination of the sentinel animal and the target species to be investigated are processes that require a detailed study to carry out an increasingly better approach to associate its risks with the species human. The use of sentinel animals ranges from the use of wild species to domestic animals, both production and companion animals, which present a series of advantages, disadvantages, and limitations that must be considered in this type of research to achieve the most accurate results that allow their extrapolation under conditions like those exposed in humans.

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Chapter 8

Advanced Oxidation Process Applied to Cyanidation Treatment of Wastewater: A Case Study of a Gold Mining Area in Colombia



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1 Introduction

Advanced oxidation processes (AOPs) have aroused great interest due to their high efficiency in the removal of potentially toxic elements (PTEs) in wastewater. The most representative types of AOPs are photocatalytic oxidation, Fenton oxidation, electrochemical oxidation, ozonation and plasma oxidation; in addition, the combination of these processes or other treatments generates hybrid technologies or treatments that stand out for their improved performance. These AOP technologies have been implemented on a large scale for treatment of drinking water and water reuse facilities; however, for the treatment of wastewater with high content of PTEs, the investigations are still at the laboratory and pilot scale. This chapter presents the results for two AOP treatments, heterogeneous photocatalysis and photo-Fenton process, at the laboratory scale. These treatments show high efficiency and great potential as a management of cyanide removal from wastewater of gold mining industry in southern Bolivar (Colombia). The results provide a basis of information and methodology to guide the application of this technology in the treatment of wastewater from mining processes.

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2 Environmental and Human Health Impacts of Cyanide in Gold Mining Activities

The environmental impacts caused by gold mining in Colombia, despite improvement in the regulations that have been applied in the producing departments, are a matter of concern for governmental and non-governmental organisations responsible for the care of the multiplicity of ecosystems available in the country (Najera et al., 2011). Within the possible consequences of this activity is the release of toxic substances that, in excess, are harmful to living beings. These hazardous wastes are produced in large volumes and their characteristics depend mainly on the process implemented for metal extraction—the most commonly used processes being amalgamation and leaching (Unidad de Planeación Minero Energética, 2007). Cyanidation treatment consists of treatment by percolation, agitation of the sands, concentrates or amalgamation residues carrying values, with a cyanide solution. This substance dissolves gold forming complexes with a reaction kinetics that is mainly affected by the exposed surface area of gold, concentration of reagents, mineral composition and time (Botero, 2021). The residues generated in gold leaching are characterised by high concentrations of free and complexed cyanides (CN^-), which must be eliminated to avoid contamination and environmental deterioration (Barakat et al., 2004).

Humans are exposed to cyanide through activities such as drinking water, touching of soil or eating food contaminated with cyanide. The composition of cyanide can determine the severity of human health effects; this could be hydrogen cyanide gas or cyanide salts. When a person is in contact with high levels of cyanide, it can cause serious brain or heart conditions, including death (Agencia Nacional de Minería [ANM], 2020).

Another property that explains the functionality of cyanide in mining and its toxicity is the ability to bind to metals such as iron (Fe), silver (Ag), gold (Au), nickel (Ni), zinc (Zn), cadmium (Cd) and mercury (Hg), among others. An example of this is the way in which cyanide affects cellular respiration by binding to the Fe atom in the centre of the structure of the enzyme cytochrome C oxidase, which is essential in this process, causing cellular asphyxia. Likewise, cyanide affects the normal flow of oxygen in the blood by binding with the Fe atom present in haemoglobin; hence, use of cyanide is considered highly risky (Hendry-Hofer et al., 2019).

In general, the use of cyanide could seriously affect human health, presenting respiratory, dermatological, cardiovascular or even cancer symptoms in affected populations (Abedi et al., 2020 as cited in Wong López, 2021). In mining, cyanide residues affect all ecological receptors: plant species, mammals, reptiles, amphibians, birds and fish, the latter being the most affected species because of their exposure to open ponds, where wastewater from the mining industry is discharged. In aquatic ecosystems, the presence of cyanide concentration of $\geq 7.2 \mu\text{g L}^{-1}$ reduces aquatic mobility and inhibit reproduction in many fish species (Logsdon et al., 2002).

In gold mining, it is common to use cyanide in the recovery processes of the material because this element provides greater efficiency compared to mercury. In addition, it allows the recovery of low-grade gold ores, since weak alkaline solutions have

a direct dissolving and preferential action on the gold contained in the ore. However, the problem of cyanide in gold mining is the lack of treatment of residual discharges and the ineffective environmental control by the authorities, which in socially vulnerable areas, such as the south of Bolivar, lack the presence and management to ensure the proper disposal of hazardous liquid waste (Díaz-Arriaga, 2014).

In Colombia, the mining sector generates 350,000 direct jobs and close to 1 million indirect jobs; this makes it one of the key sectors for the country's economic reactivation (Agencia Nacional de Minería [ANM], 2020). Mining companies contributed \$518 billion pesos in royalties during the first quarter of 2020 and close to 5 trillion pesos between 2018 and 2019 because of the fact that the contributions of minerals such as gold and nickel increased by 38% and 40%, respectively, confirming the sector's progress in mining diversification (ANM, 2020).

According to the information drawn from ANM (2017), the department of Bolivar has 377 mining titles covering an area of 310,722 ha, with 68 titles for mining exploration, 93 in construction and assembly, and 216 in exploitation, making this economic activity as one of the most dynamic economic lines that support incomes of the population. In addition, the proximity of these projects to the Mojana region plays a fundamental role because of its environmental importance as an integral part of the Momposina depression wetlands complex, serving as a buffer for flooding and distribution of flows caused by rainfall in the highlands of the Andean region (Montenegro and Gutiérrez, 2013).

Thus, in the area, there is a great socio-environmental problem due to the precarious technological conditions with which the mining activity is carried out, the excessive use of mercury and cyanide in gold extraction, the proximity of mines to the main sources of surface water, the lack of organisation of producers, the weakness of strategic plans that seek the implementation of treatment technologies, and the low level of environmental awareness of miners that still prevails in this region of the country (Padilla Camacho et al., 2005; Rey Quijano, 2008). In this sense, innovative proposals are needed to establish solutions to the socio-environmental crisis in this region of the country.

3 Advanced Oxidation Processes in Environmental Remediation

Established methods for the treatment of cyanide-contaminated effluents include physical methods, such as dilution, filtration, adsorption with activated carbon; chemical methods, such as alkaline chlorination and oxidation using hydrogen peroxide (H_2O_2) and ozone (O_3); physicochemical methods, such as ion exchange and electrolysis; and biological methods, such as biodegradation (Morillo Esparza et al., 2019). However, these conventional methods have certain limitations. Physical treatments cannot be applied on a large scale and at high wastewater concentration levels, and chemical methods require expensive processes and generate toxic by-products

(Augugliaro et al., 1999; Osathaphan et al., 2008). For instance, alkaline chlorination, widely used in the mining industry, generates trihalomethanes and cyanogen chloride, which are highly carcinogenic (Malhotra et al., 2005). Also, biological treatments are not effective in the case of large quantities and high concentrations of wastewater and require a long-term process.

In recent decades, AOPs have established themselves as an efficient alternative for the removal of toxic substances, including organic pollutants (phenols, surfactants, oils and greases), inorganic pollutants (chlorides and nitrates), heavy metals (arsenic, mercury and lead), among others (Elmobarak et al., 2021).

These AOPs are composed of a varied and wide range of technologies based on the generation of hydroxyl radicals ($\text{HO}^\bullet$) that degrade most organic compounds (Sanz et al., 2012; Fernández-Castro et al., 2015). AOPs can be used to treat effluents from different sources such as oil, metal and agrochemical industries; wastewater treatment plants with the purification of infectious-contagious organisms; and effluents from health centres and animal slaughter plants (Escudero Santiago, 2017).

Advanced oxidation processes result in the transformation of contaminants, generally without producing sludge; they are effective and useful in the removal of refractory contaminants that require other treatment methods, especially the biological method. They allow the treatment of contaminants found in low concentrations; the contaminants generated in the processes are biodegradable in the water treated with conventional biological processes; they contribute to the improvement of the organoleptic conditions of the treated water (Domènech et al., 2001; Pinos, 2020).

3.1 *Heterogeneous Photocatalysis*

Heterogeneous photocatalysis is a process based on the direct or indirect absorption of radiant energy (visible or ultraviolet [UV]) by a solid, which is usually a wide band-gap semiconductor. Figure 1 depicts the process occurring at the semiconductor–electrolyte interface under UV illumination. Once the semiconductor particle is excited with sufficiently energetic light ($254 \text{ nm} < \lambda < 310 \text{ nm}$), electron-hole pairs are created whose half-life is in the range of nanoseconds; in this time, they must migrate to the surface and react with the adsorbed species (process c and d in Fig. 1). Electron-hole pairs that do not react with species on the surface recombine and energy is dissipated. This recombination can take place both on the surface and within the particle (process a and b in Fig. 1) (Domènech et al., 2001).

The most investigated photocatalysts so far are the wide band-gap semiconductor metal oxides, and in particular, titanium dioxide (TiO_2), which has a high chemical stability that makes it suitable to work in a wide pH range (Garcés et al., 2004). In addition, TiO_2 has a refractive index of 2.4 and is one of the chemical substances with the capacity to reflect practically all visible radiations that reaches it, thus maintaining its colour permanently (Rodríguez, 2011). The photodegradation reactions of cyanide are shown in Equations (1) and (2) (Augugliaro et al., 1999; Correa Jaramillo, 2006):

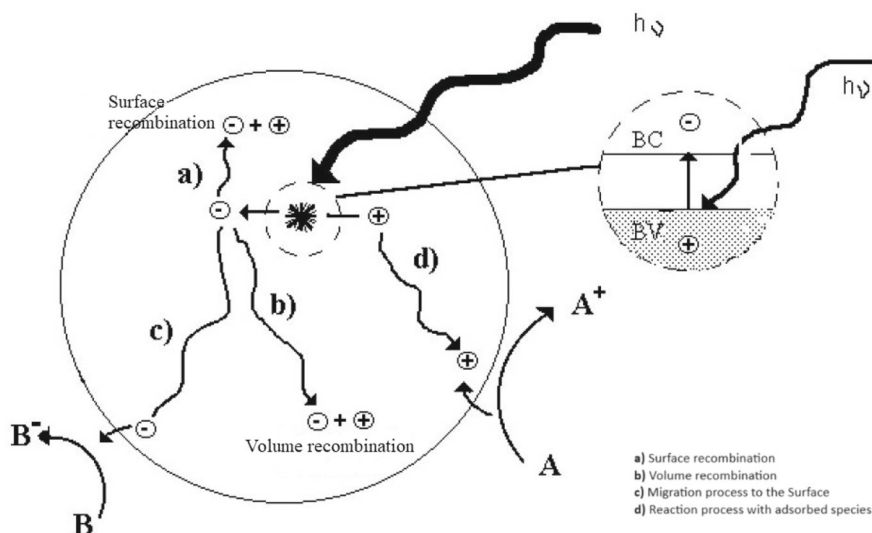
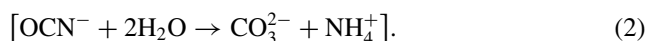
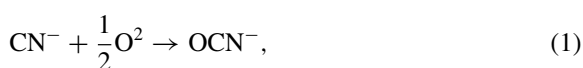


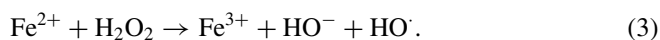
Fig. 1 Mechanism of semiconductor action. (Adapted from Domènech et al., 2001.)



Different parameters influence the photocatalytic treatment, such as the following: wavelength, which must be less than 400 nm (nanometer) so that it can be absorbed by TiO_2 ; concentration of the catalyst, which can increase turbidity by decreasing the effect of UV light; dissolved oxygen, because it acts as an electron acceptor, generating the superoxide radical; and finally, temperature and pH that can interfere minimally in the speed of the reactions, for example, at neutral pH, the particle size and surface charge are affected. For this reason, it is recommended to work at a pH different from the isoelectric point for TiO_2 (pH = 7) (Garcés et al., 2004, 2006).

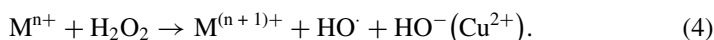
3.2 Photo-Fenton Process

This AOP is based on the combination of Fe salts and H_2O_2 under UV light, which upon reaction generates highly oxidizing $\text{HO}^\cdot$. Photo-Fenton process has been tested successfully for removing toxic compounds, such as textile dyes, halomethanes and chlorinated organic compounds, among others. Equation (3) corresponds to the reaction that occurs during this process:

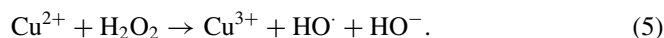


The process should be carried out at a pH of about 2.8 to operate in optimal conditions, otherwise the treatment efficiency could be decreased, because at $\text{pH} > 3.0$, Fe precipitates, making the degradation reactions of the pollutant impossible (Parag and Aniruddha, 2004). However, for safety reasons, it is advisable not to work with cyanide at acidic conditions because of the generation of gaseous HCN, which is of high risk for human health. It is recommended to maintain pH in a range of 10 to 11 (Noya et al., 2011). This condition represents an initial disadvantage for the application of this AOP.

Photo-Fenton process has emerged as an alternative to cyanide treatment. Fe salts are replaced by other metals that exert their photocatalytic function favouring the generation of $\text{HO}^\cdot$. The most commonly used transition metal is divalent copper (Cu^{2+}). The general reaction of a photo-Fenton process is shown in Equation (4):



The photo-Fenton reaction using Cu^{2+} as a transition metal is represented in Equation (5):



4 Study Area and Related Problems

The water samples were collected through the cyanidation process neutralisation tailings of a gold mine located in the south of the Bolivar department, with the following coordinates: latitude, North $08^\circ 56' 20.3''$ and longitude, West $74^\circ 02' 42''$. The work carried out in the mining project involves the exploitation of the deposit and gold beneficiation.

The experiment was carried out with water doped with sodium cyanide (NaCN) having 95% purity, due to the difficulty of transporting large volume of water from mine to laboratory. After optimisation of the treatment, three replicates were carried out with the mine water sample for verifying the efficiency of the process with real sample.

5 Methodology

An 8-h sampling was carried out directly at the discharge point. Samples were collected in 20-L dark plastic tanks, previously washed and dried for transport to the Toxicology and Environmental Management Laboratory of the University of Córdoba.

Physicochemical parameters were evaluated for characterisation of the samples based on the procedures established by the American Public Health Association (APHA) (Lipps et al., 2023), and determining the initial and final concentration of cyanide was important. The characterisation was taken as a reference point to calculate cyanide removal percentage and evaluate photocatalytic treatment.

5.1 Treatment System in Place

5.1.1 Heterogeneous Photocatalysis with TiO_2

Preliminary tests were carried out prior to the development of the experimental design in order to verify the information obtained from literature reviews and to strengthen the research by exploring different possibilities and variables that made it possible to establish ranges to evaluate each of the factors.

A Mygthy Pure Ultraviolet Water Purifier MP22A photo-reactor with a 3.8-L capacity was employed. The external cylinder was made of stainless steel and a quartz tube inside it contained a low-pressure mercury lamp with power equal to 20 W that reaches up to 254-nm wavelength.

The catalyst used was TiO_2 Degussa P-25 (80% anatase and 20% rutile), with a high catalytic activity when exposed to 254–310-nm radiation (Villanueva et al., 2004). The photo-reactor had its respective suction and discharge connections to direct the solution to be treated to a 4-L container completely lined to avoid penetration of sunlight; a submersible motor pump model 2×10^{-38} N with a flow rate equal to $22.709 \text{ L min}^{-1}$ was placed in this container. The container was filled with 3.2-L cyanide water, completely submerging the pump and obtaining a total of 7-L water to be treated, which was kept in recirculation for 2 h.

5.1.2 Photo-Fenton Process

The above-mentioned Mygthy photo-reactor was used for heterogeneous photocatalysis. The samples to be treated were added together with hydrogen peroxide and copper sulfate ($\text{H}_2\text{O}_2\text{--CuSO}_4$), the solution was set at $\text{pH} > 9.8$ to avoid formation of gaseous HCN, and irradiated for 1 h.

5.2 Experimental Design and Treatment Results

5.2.1 Experimental Design of Heterogeneous Photocatalysis with TiO_2

Heterogeneous photocatalysis was performed applying the Box–Behnken experimental design with three factors and three levels (Table 1); this is one of the

Table 1 Heterogeneous photocatalysis experimental design factors and levels

Independent variable	<i>Factor</i>	<i>Rrnk rnd level</i>		
		<i>−1</i>	<i>0</i>	<i>1</i>
[CN [−]] (mg L ^{−1})	A	30	50	100
[TiO ₂] (g L ^{−1})	B	0.01	0.05	0.1
[H ₂ O ₂] (mL L ^{−1})	C	1	2	4

experimental designs used to adjust response surfaces (Montgomery, 2004). The response variable was cyanide-removal percentage. The applied design comprised 15 experimental units and three central points.

Appropriate concentrations of TiO₂ and H₂O₂ were added to the sample to be treated, and pH of the solution was set at >9.8 in order to avoid the formation of gaseous HCN in the experimental phase of the research. The solution was stirred for 20 min in the dark to reach adsorption–desorption equilibrium on the catalyst surface; after 20 min, the lamps were turned on to start the treatment. The Cyanide-Selective Electrode Method (SM: 4500-CN-F) (Lipps et al., 2023) was used to measure initial and final cyanide concentrations.

5.2.2 TiO₂ Recovery and Reuse

The sedimentation process, which consisted of a 20-L sedimentation tank, with inlet and outlet connections, was implemented as a post-treatment to clear up the treated water. To determine TiO₂ sedimentation time, tests were performed using an Imhoff cone, without using any type of coagulant or varying the pH conditions. The settler was conditioned with an extra outlet tap at the bottom to discharge concentrated TiO₂ solution that remained at the bottom, and was stored for reuse. Figure 2 shows the design of the employed photocatalytic treatment system.

The reuse of catalyst was carried out based on the methodologies described by Fernández (2003). Firstly, the TiO₂ collected by the sedimentation process was dried to a solid state in order to reincorporate it into a new photocatalytic treatment as shown in Fig. 3. Secondly, it could be possible to reuse the concentrated solution of catalyst without drying it. This could be achieved through the calculation of a calibration curve that relates turbidity to TiO₂ concentration. This would possibly estimate the amount of TiO₂ poured by measuring the turbidity of the clarified water in order to add it to a new treatment together with the concentrated TiO₂ solution.

5.3 Experimental Design of Photo-Fenton Process

The photo-Fenton process was performed applying the Box–Behnken experimental design with three factors and three levels. This design consisted of 15 experimental

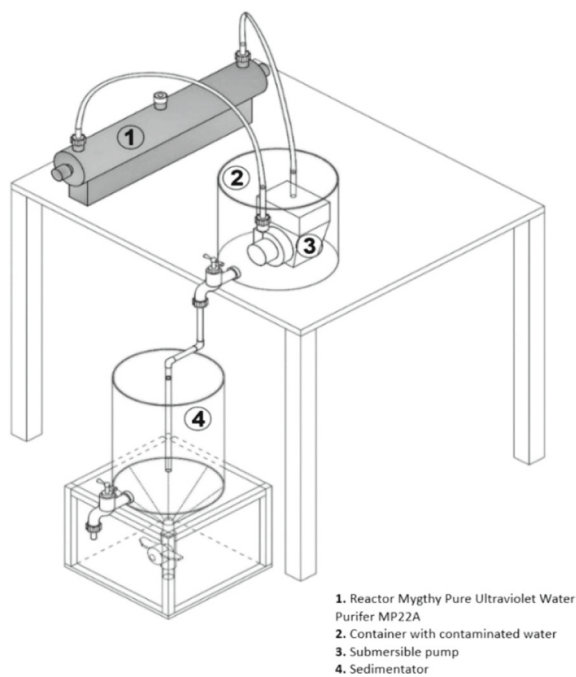


Fig. 2 Photocatalytic system design. (Source: the present study.)

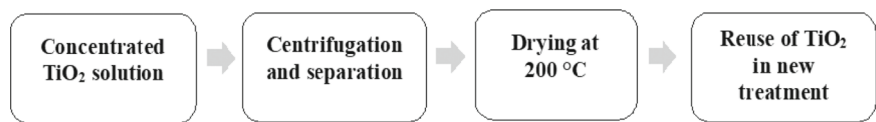


Fig. 3 Separation and TiO₂ drying. (Source: the present study.)

units and three central points as shown in Table 2. This way, the efficiency of both photocatalytic treatments in cyanide removal was compared.

After the reaction time, the treated solution was transferred to a settling tank to clear up the water before discharge. The resulting precipitate was dried, stored, labeled and delivered to a hazardous waste manager, who was responsible

Table 2 Photo-Fenton factors and levels of experimental design

Independent variable	Factor	Rank and level		
		-1	0	1
[CN ⁻] (mg L ⁻¹)	A	30	50	100
[CuSO ₄] (mg L ⁻¹)	B	10	15	20
[H ₂ O ₂] (mM)	C	20	40	60

for its collection, transport, treatment and final disposal in accordance with the environmental regulations applicable to this type of waste.

5.3.1 Processing of Results

In order to determine the effectiveness of the system in the removal of NaCN, the percentage of removal was calculated using Equation (6):

$$\text{Removal(\%)} = \frac{C_i - C_f}{C_i} \times 100, \quad (6)$$

where C_i is the initial cyanide concentration and C_f corresponds to the final cyanide concentration.

The response surface methodology (RSM) statistical analysis was applied to evaluate the single and combined effects of independent parameters [TiO_2], [H_2O_2], [CN^-] and [CuSO_4] in cyanide removal, and optimise the conditions of each photocatalytic treatment (Jaramillo et al., 2013). To evaluate significant differences between the results, the analysis of variance (ANOVA) was performed. The significance criterion was set at $p < 0.05$.

Process optimisation was based on studying the response of statistically designed combinations that best fitted the experimental conditions. From these values, experimental tests were carried out using the sample from the neutralisation tails of the cyanidation process of gold mine, and the reaction kinetics were calculated.

6 Results

6.1 Physicochemical Characterisation of Water Cyanidation

Table 3 presents the physicochemical characterisation of the sample from the neutralisation tails of a gold beneficiation process located in the south of the department of Bolívar.

These results show a high concentration of cyanide that exceeded the limit of 1 mg L^{-1} , established in Article 10 of Resolution 0631 of 2015 for gold and other precious metal extraction activities, making it necessary to treat these water samples for

Table 3 Physicochemical characterisation of the sample

Parámetro	Unidad	Método	Resultado muestra	Resolución 0631 de 2015
pH	NA	SM: 4500-H +B	11.40	6.00-9.00
Cianuro	mgL^{-1}	SM: 4500-CN-F	78.77	1.00

removal of cyanide. Water pH also presented high values that exceeded those established in the resolution. However, this result was favourable when performing photocatalytic treatments, which must be carried out at $\text{pH} > 9.5$ to avoid the formation of HCN (Barbosa López and Castro Sierra, 2012).

6.2 Implementation of Pilot Treatment

The results of the 15 tests corresponding to the proposed Box–Behnken-type experimental design are shown in Table 4. The results showed that 73% of the tests performed complied with the provisions of Resolution 0631 of 2015 regarding the maximum permissible limit for cyanide. The minimum value for cyanide removal was 68.8%, with a maximum of 99.9% and an average of 93.9%, which established a higher trend of data towards the maximum value of cyanide removal.

Table 4 Results of heterogeneous photocatalysis

Test	$[\text{TiO}_2]$	$[\text{H}_2\text{O}_2]$	Theoretical $[\text{CN}^-]$	Experimental $[\text{CN}^-]$	Final $[\text{CN}^-]$	Cyanide removal (%)
1	0.10	4	50	49.7	0.320	98.603
2	0.01	1	50	49.2	0.327	98.965
3	0.05	4	100	97.7	0.689	98.476
4	0.05	4	30	33.8	0.101	98.196
5	0.01	2	100	90.9	0.034	99.941
6	0.05	1	30	36.3	0.471	98.471
7	0.05	2	50	46.1	0.215	99.311
8	0.10	1	50	49.1	8.210	77.811
9	0.05	1	100	101.0	17.900	77.956
10	0.01	2	30	36.0	0.025	99.874
11	0.10	2	30	27.9	0.033	99.750
12	0.05	2	50	45.0	0.115	99.615
13	0.01	4	50	45.6	0.196	99.018
14	0.10	2	100	75.4	17.100	69.841
15	0.05	2	50	61.0	3.470	93.327

6.3 Optimisation of Heterogeneous Photocatalysis Conditions

Pareto diagram (Fig. 4) enabled the analysis and identification of factors. The ANOVA analysis identified the most significant factors in cyanide removal, with each factor represented in horizontal bars. Thus, the factors that exceeded the vertical line were considered influential in the degradation of the pollutant. According to this graph, TiO_2 concentration was considered as the factor with the highest statistical significance. However, the negative sign (-) indicated an inverse proportional relationship, where lower concentrations of TiO_2 influenced higher values of the response variable. This was justified because the catalyst considerably increased the turbidity of water, reducing the action, and reaching UV radiation, as shown in Fig. 5. In the same way, H_2O_2 concentration had a direct proportional relationship and showed a great incidence of cyanide degradation.

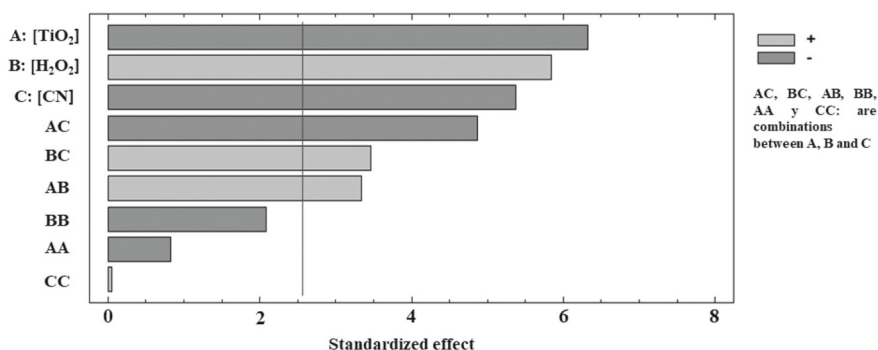


Fig. 4 Pareto diagram for heterogeneous photocatalysis. (Source: the present study.)

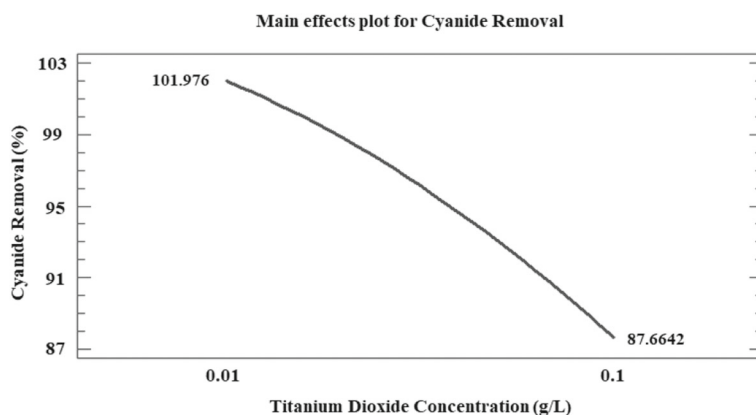


Fig. 5 Effect of TiO_2 concentration on cyanide removal. (Source: the present study.)

The coefficient of determination (r^2) for this treatment was 0.9678 (97%). However, the adjusted r^2 statistic, which is more appropriate for comparing models with a different number of independent variables, was 0.9100 (91%), implying that only 9% of variations in cyanide removal could not be explained by the model. Taking into account that Montgomery (2004) suggested that the coefficient of determination should be at least 75%, it was possible to optimise response variable from RSM analysis.

Equation 7 presents the second-order regression model showing the relationship between the percentage of cyanide removal and the previously defined variables:

$$\begin{aligned} \text{Cynade removal} = & 102.04 + 20.06x + 2.65y - 0.16z - 652.86x^2 \\ & + 72.15xy - 4.43xz - 1.66y^2 + 0.09yz + 7.76 \times 10^{-5}z^2, \end{aligned} \tag{7}$$

where x , y and z represent TiO_2 , H_2O_2 and CN^- concentrations, respectively.

The estimated response surface grid (Fig. 6) allowed visualising the performance of response variable against the level combination of different factors evaluated in the design. It indicated that the highest cyanide removal was observed in orange- and red-shaded areas, with a removal proportion of 95–100%.

Based on the model indicated, the optimal treatment conditions (Table 5) found by means of statistical software should result in 100% cyanide degradation.

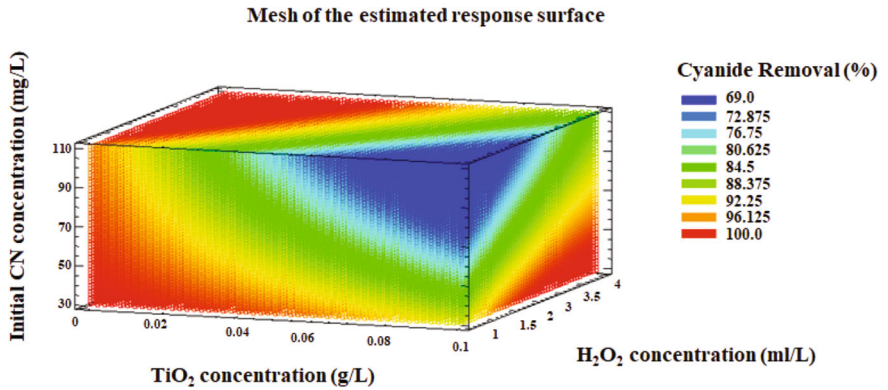


Fig. 6 Estimated response surface of heterogeneous photocatalysis. (Source: the present study)

Table 5 Optimisation of heterogeneous photocatalysis

Factor	Low	High	Optimal
[TiO ₂]	0.01	0.1	0.0364045
[H ₂ O ₂]	1.0	4.0	2.99454
[CN ⁻]	30.0	100.0	63.1592

Three replicate tests were carried out with real samples, using the concentrations obtained in Table 5 for $[\text{TiO}_2]$ and $[\text{H}_2\text{O}_2]$ and maintaining initial cyanide concentrations obtained in physicochemical characterisation (Table 3; Rodríguez Santiago, 2013). As a result, an average cyanide removal rate of almost 99% was obtained, which demonstrated the high effectiveness of treatment in the degradation of this pollutant and satisfied the predictions of the model.

6.4 Comparison Between Implemented Pilots

The results of 15 samples corresponding to the Box–Behnken experimental design for photo-Fenton process are presented in Table 6.

Only 26.7% of the tests carried out were able to reduce cyanide concentration below 1 mg L^{-1} . However, the average cyanide removal was 95.4%, indicating that it was a competent alternative in the treatment of this type of water and consequently in mitigating the environmental impacts of mining activities.

The RSM statistical analysis was optimised to obtain a second-order regression model (Equation 8) showing a relationship between the percentage of cyanide removal and the previously defined variables:

$$\text{Cynade removal} = 98.79 + 1.17v + 0.37y - 0.18z + 0.05v^2 - 9.25 \times 10^{-4}vy$$

Table 6 Photo-Fenton process results

Test	$\text{CuSO}_4 \text{ (mg L}^{-1}\text{)}$	$\text{H}_2\text{O}_2 \text{ (Mm)}$	$\text{CN}^- \text{ (mg L}^{-1}\text{)}$	$[\text{CN}^-] \text{ initial}$	$[\text{CN}^-] \text{ final}$	$[\text{CN}^-] \text{ removal (}\%)$
1	20	60	50	48.8	0.184	99.63
2	10	20	50	49.3	4.690	90.49
3	15	60	100	98.6	5.120	94.81
4	15	60	30	27.2	0.240	99.12
5	10	40	100	98.7	5.620	94.31
6	15	20	30	28.9	2.080	92.81
7	15	40	50	48.8	2.780	94.31
8	20	20	50	48.6	3.400	93.01
9	15	20	100	98.6	6.320	93.60
10	10	40	30	30.1	0.311	98.97
11	20	40	30	30.1	0.272	99.10
12	15	40	50	47.9	2.350	95.10
13	10	60	50	49.1	1.240	97.48
14	20	40	100	98.2	5.240	94.67
15	15	40	50	49.0	2.920	94.05

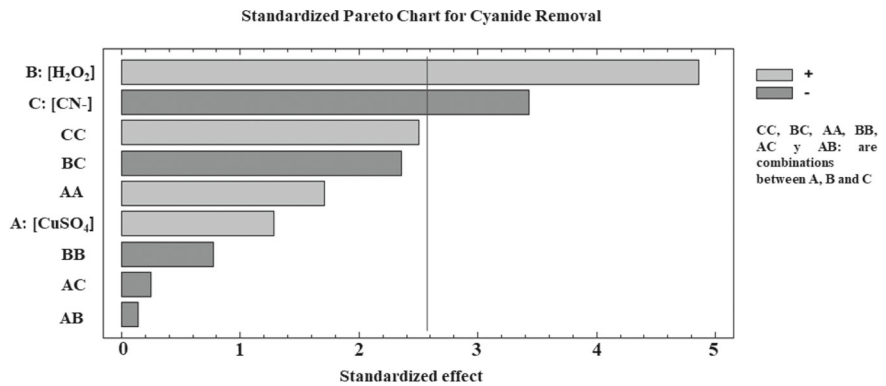


Fig. 7 Pareto diagram for photo-Fenton process. (Source: the present study.)

$$- 8.71 \times 10^{-4}vz - 1.31 \times 10^{-3}y^2 - 2.10 \times 10^{-3}yz + 1.76 \times 10^{-3}z^2, \tag{8}$$

where v , y and z represent CuSO_4 , H_2O_2 and CN^- concentrations, respectively.

The ANOVA analysis of variance established that the concentrations of independent variables, H_2O_2 and initial CN^- , were significantly different from zero at 95% confidence level, as shown in Pareto diagram (Fig. 7). In this case, H_2O_2 was the most significant factor due to its high oxidative potential, presenting a direct proportional relationship to response variable.

By contrast, CuSO_4 showed $p > 0.05$, indicating that there was no statistically significant difference between the concentrations used in the experimental design, because the cyanide removal was not affected by its variation. However, Fig. 8 presents the effect of CuSO_4 concentration on response variable, and high cyanide removal was observed.

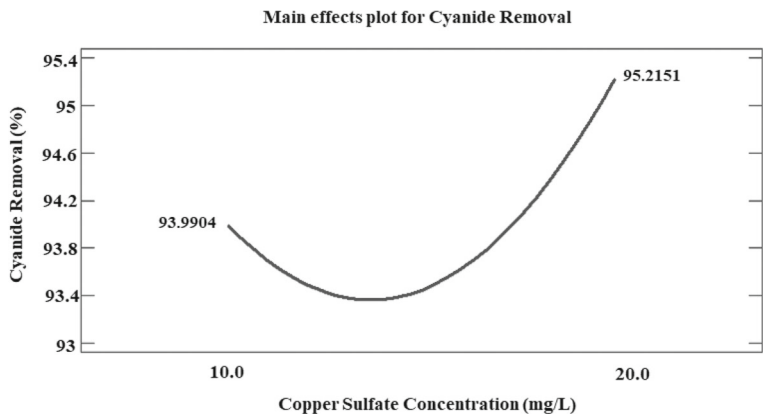


Fig. 8 Effect of CuSO_4 concentration on cyanide removal. (Source: the present study.)

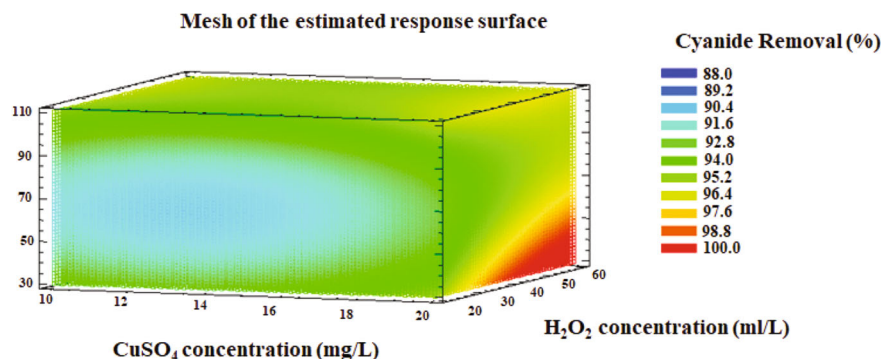


Fig. 9 Response surface grid for photo-Fenton process. (*Source:* the present study.)

Table 7 Optimal operating conditions of photo-Fenton process

<i>Factor</i>	<i>Low</i>	<i>High</i>	<i>Optimal</i>
[CuSO ₄]	10	20	14.7004
[H ₂ O ₂]	20	60	57.7228
[CN ⁻]	30	100	30.4808

The coefficient of determination (r^2) was 0.9213 (92%); however, the adjusted r^2 statistic, which evaluates models with different independent variables, was 0.7797 (78%). Therefore, 22% of variation in cyanide removal, a higher percentage compared to heterogeneous photocatalysis, could not be explained by the model, but it satisfied the criteria established by Montgomery (2004).

The estimated response surface grid (Fig. 9) showed the behaviour of response variable versus combination of levels of different factors evaluated in the study design. It showed that the highest cyanide removal was observed in yellow-, orange- and red-shaded areas, with a removal proportion of 95–100%. It should be noted that in comparison to heterogeneous photocatalysis response surface grid, the red area covered less space.

Optimal conditions for photo-Fenton process are shown in Table 7, which, according to the model (Equation 7), should result in 100% cyanide degradation. However, an average removal rate of 94.4% was obtained in three repetitions with a real sample. This was explained by the calculated $r^2 = 78\%$, which was close to the established criterion of 75%.

7 Conclusions

The removal of cyanide from a mining effluent of cyanidation process implemented in a gold mine in southern Bolivar was evaluated using heterogeneous photocatalysis and photo-Fenton process. Satisfactory results were obtained in both cases, with

removal proportions of 98.8% and 94.4%, respectively. However, only in heterogeneous photocatalysis, the final concentration of the sample satisfied the criteria established in the Article 10 of Resolution 0631 of 2015.

Treatment optimisation made possible to identify ideal conditions to carry out the process. TiO_2 concentration was inversely proportional to cyanide removal. H_2O_2 showed the highest incidence in the photo-Fenton process due to its excellent oxidative capacity that increased speed of the reactions. Cyanide removal behaviour versus temperature indicated that this factor did not accelerate the rate of photocatalytic reactions. In addition, through a sedimentation process, it was possible to recover high proportion of TiO_2 , which had an efficient possibility of reuse. Thus, photocatalytic treatment is presented as an alternative to mitigate the environmental impacts caused by the discharge of cyanide wastewater from gold mining activities.

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Chapter 9

Hydrogeochemical Characterization of Groundwater in Ambagarh Regions: Mining-Rich State Chhattisgarh, India



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1 Introduction

Groundwater is a vital natural resource for a healthy society and economy. The study of groundwater involves both geological and chemical aspects and its importance in characterizing the natural systems, understanding the contaminant sources involved, and their remedies. Several authors have discussed the hydrogeochemical studies of groundwater (Lawrence et al., 1976; Soulsby et al., 1998; Wen et al., 2005; Ravikumar et al., 2010; Pradhan and Biswal, 2018; Behera et al., 2019; Pradhan et al., 2022) and hydrogeochemical assessment through bivariate and multivariate statistical studies (Dalton and Upchurch, 1978; Reghunath et al., 2002; Cloutier et al., 2008; Pradhan et al., 2021). All geochemical processes and reactions are initiated once the rain-water percolates into the subsurface. Further, the geochemistry of groundwater in the

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aquifer system largely depends upon mineral constituents (Lakshmanan et al., 2003; Pradhan et al., 2022). The groundwater movement from recharge to discharge areas alters the concentration of chemical species (Freeze and Cherry, 1979; Pradhan et al., 2022). The hydrochemistry and aquifer properties of groundwater vary temporarily and spatially depending upon their geology and chemical characteristics. Several studies have investigated the groundwater in different aquifer systems over space and time, which is vital for solving different geochemical problems (Gibbs, 1970; Behera et al., 2019; Pradhan et al., 2022). The present chapter is therefore aimed at the evaluation of hydrogeochemical characterization of groundwater and its suitability in a subtropical region of the Ambagarh areas (Rajnandgaon district), Chhattisgarh, India. This also includes the ionic concentration, ion-exchange processes, geochemical relationships and suitability for domestic and irrigation purposes which can be optimized for management of water resources.

2 Study Area and Geological Settings

The study site (Ambagarh) is located in the Rajnandgaon district of Chhattisgarh state and it lies between latitude 20°39'35" N to 20°51'00" N and longitude 80°38'50" E to 80°49'28" E (Fig. 1). The study site covered an area of about 330 km² and was in subtropical climate. Geologically, the area comes under the Central India Craton and is part of the Dongargarh–Kotri rift zone. The area is surrounded by Paleoproterozoic Dongargarh granites in the western section, Mesoproterozoic platform sequence of the Chhattisgarh

supergroup in the north-east section, and Paleoproterozoic supra-crustal sediments in the southern section (Acharyya et al., 2005; Mukherjee et al., 2009; Mukherjee and Ray, 2010). Further, the study site is situated at the north of the regional water body that separates the Mahanadi and Godavari river basins. Seonath is the major perennial tributary that shows a sub-dendritic drainage pattern. High drainage density is mostly confined in volcanic rock areas, especially in the southern part, and poor to moderate in granitic and lateritic capping areas. Rhyolites hold predominant rock types in this region. Groundwater in the study region mostly occurs in phreatic and semi-confined conditions, which are more potential and productive than the granitic rocks.

3 Materials and Methods

In order to characterize the hydrogeochemical and physicochemical studies of groundwater, 30 representative groundwater samples were used around the Ambagarh areas of Rajnandgaon district (Fig. 1). Table 1 shows the in-situ and chemical parameters of groundwater samples. The results of water samples were evaluated

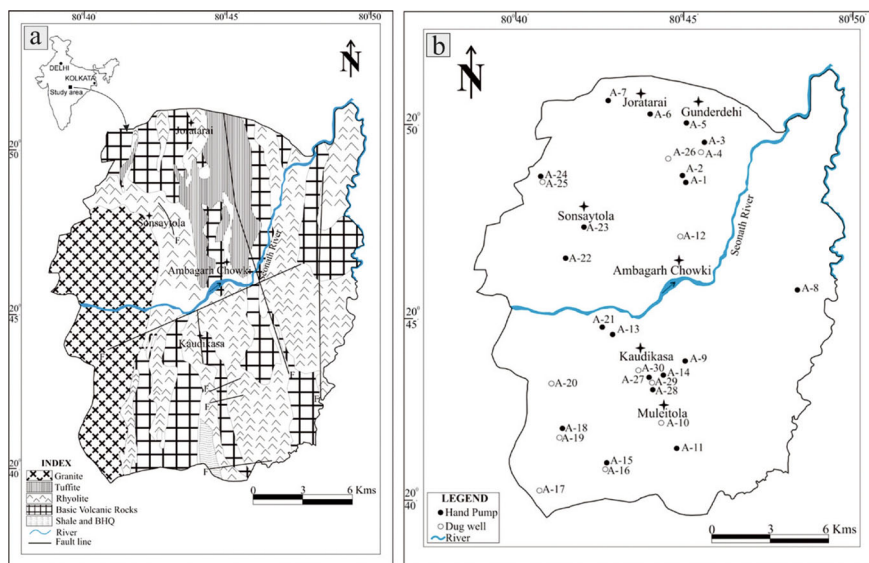


Fig. 1 (a) Lithological map of the study area (Mukherjee et al., 2009) and (b) groundwater sample locations.

following the drinking water quality standards recommended by the American Public Health Association (APHA, 1992) and the World Health Organization (WHO, 1993).

4 Results and Discussion

4.1 General Hydrogeochemistry and Hydrogeochemical Facies

The value of electrical conductivity (EC) ranges from 182 to 1,388 $\mu\text{S}/\text{cm}$ whereas the pH varies from 7.6 to 8.2 (Table 1). Table 2 shows the maximum and minimum concentrations of major ions analyzed for the present study. Various graphical methods (Zaporozec, 1972; Güler et al., 2002) are used to classify and visualize the hydrochemical data of groundwater samples from their chemical compositions. Based on the ionic composition, the most widely accepted Piper trilinear plot is used to determine the possible source of dissolved constituents in the groundwater along its flow path (Piper, 1944). Further, the hydrochemical facies are largely a function of different lithology, groundwater flow patterns, and solution kinetics. The concentration of major ions in the groundwater varies spatiotemporally, showing mostly Ca-HCO_3 and Ca-Cl-HCO_3 water types (Fig. 2). The Ca-HCO_3 water type is possibly caused due to rainfall recharge processes, with low EC values. On the other

Table 1 Major ion concentrations in groundwater samples in Ambagarh areas

Sample	pH	EC	TDS	Ca	Mg	Na	K	HCO ₃	Cl	SO ₄	As
		(μ S/cm)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(μ g/L)
A-1	7.7	428	299.6	30	17	35	0.6	232	1	20	ND
A-2	7.8	353	247.1	30	15	27	0.7	195	11	12	ND
A-3	8.0	1,198	838.6	80	41	95	12	134	152	200	ND
A-4	7.7	261	182.7	28	9	15	0.8	140	11	ND	9
A-5	7.6	201	140.7	18	6	16	0.5	92	7	0	120
A-6	8.1	269	188.3	26	9	25	0.3	128	14	0	32
A-7	7.9	988	691.6	76	24	91	0.9	140	135	165	ND
A-8	7.9	686	480.2	62	13	58	0.8	220	39	65	ND
A-9	7.7	407	284.9	28	17	29	0.6	189	14	30	61
A-10	7.8	715	500.5	66	30	31	0.6	201	57	62	9
A-11	7.7	615	430.5	30	30	57	1	256	35	52	ND
A-12	7.7	349	244.3	28	13	26	1.1	49	43	30	ND
A-13	7.6	418	292.6	40	16	19	0.7	165	35	24	ND
A-14	7.8	687	480.9	66	16	52	0.7	146	74	106	ND
A-15	7.8	605	423.5	18	43	43	0.9	171	53	67	ND
A-16	7.9	745	521.5	64	13	84	1	171	74	130	ND
A-17	7.7	468	327.6	42	16	28	0.4	207	18	34	ND
A-18	7.7	680	476	76	26	28	0.9	226	85	38	10
A-19	7.6	250	175	24	9	16	1	110	14	14	ND
A-20	8.1	690	483	66	23	28	28	183	53	62	ND
A-21	7.8	680	476	58	6	85	1.1	220	64	62	ND
A-22	8.0	182	127.4	18	1	19	0.6	79	7	ND	ND
A-23	7.7	527	368.9	56	9	53	0.6	85	57	72	115
A-24	7.9	1,113	779.1	92	58	40	0.7	281	43	205	ND
A-25	8.1	863	604.1	34	50	66	5.3	146	167	48	ND
A-26	8.2	1,330	931	24	33	150	1203	262	113	230	ND
A-27	8.0	689	482.3	74	16	39	1.5	104	85	77	18
A-28	7.9	942	659.4	58	54	42	13	201	92	145	ND
A-29	8.1	608	425.6	42	5	79	10	171	39	86	490
A-30	7.8	1,388	971.6	56	6	100	84	226	152	225	8

hand, when the recharge process is not prominent, the concentration of chloride and sodium increases, leading to the Ca-Cl-HCO₃ water type. Most of the samples in this region are bicarbonate types whereas in some cases, chloride and sulphate ions dominate mostly.

Table 2 Minimum and maximum of electrical conductivity (EC) and major ion concentrations

Parameters	Minimum	Maximum	Unit
EC	182	1,388	μS/cm
Ca	18	92	mg/L
Mg	6	58	mg/L
Na	15	150	mg/L
K	0.3	120	mg/L
HCO ₃	49	281	mg/L
Cl	7	167	mg/L
SO ₄	0	225	mg/L

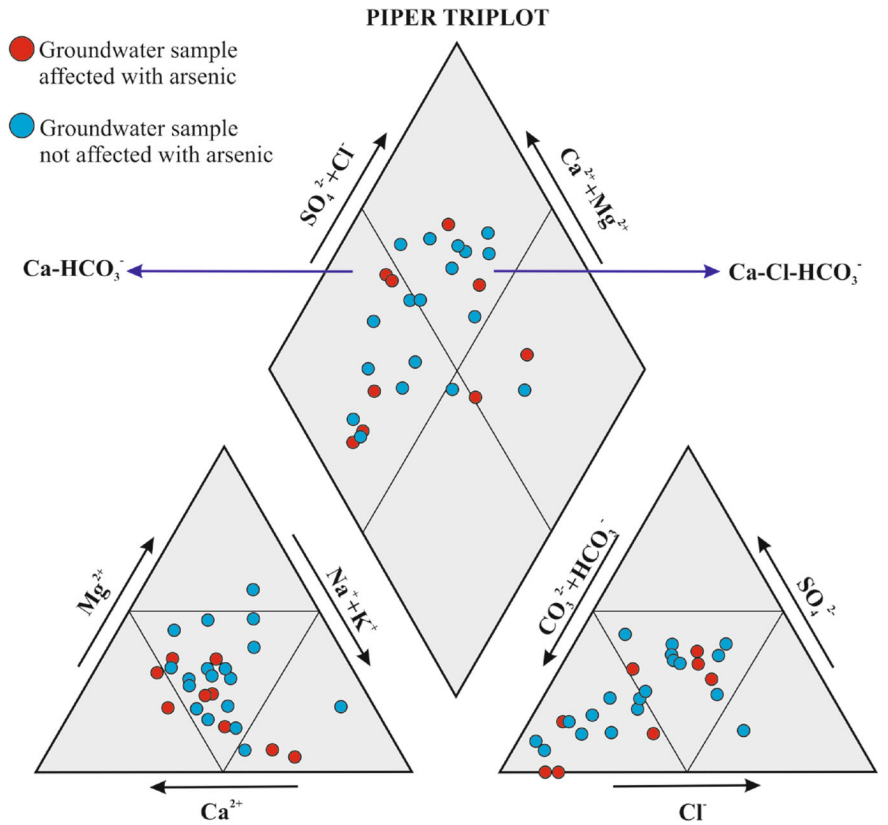


Fig. 2 Piper plot showing major groundwater types.

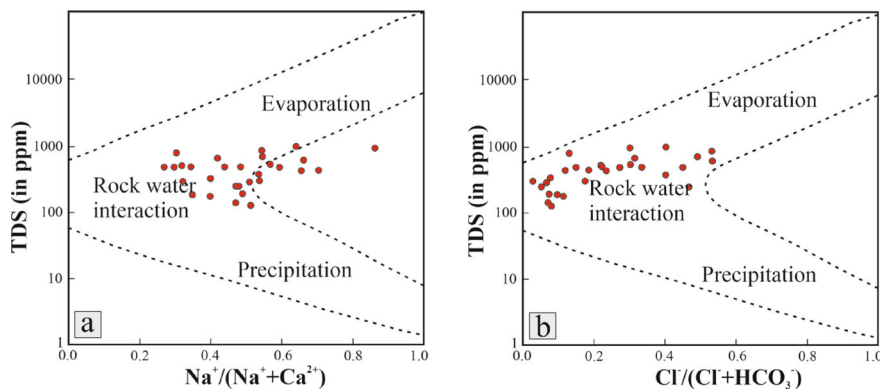


Fig. 3 Gibbs (1970) plot showing rock–water interaction mechanisms.

4.2 Rock–Water Interaction Mechanisms

The Gibbs plot in Fig. 3 shows the mechanisms that control the chemical relationships of groundwater based on the aquifer that distinguishes the impacts of evaporation, precipitation and water–rock interaction. The groundwater samples in the study area fall in the central portion and upper part of the Gibbs plot, suggesting the weathering of rock-forming minerals and evaporation dominance process, respectively (Fig. 3). Most of the groundwater falling in the water–rock interaction region indicates that the dissolved ions are derived from the host rocks, while a small number of samples belong to the evaporation region, suggesting high temperature prevailing in the study area which results in evaporation rate.

4.3 Hydrogeochemical Processes

In the groundwater system, various geochemical processes contribute to chemical variation in groundwater which can be described by bivariate plots (Fig. 4). In Fig. 4a, the scatter plot of total cation versus alkali metals (Na+K) shows that nearly all the groundwater samples lie below the 2:1 reference line which indicates the presence of a higher concentration of alkali earth minerals than alkali minerals. The study area is mainly covered by rhyolite and the volcanic equivalent of granitic magma, which essentially comprises quartz, K-feldspar and biotite minerals. The rainwater (usually Ca-HCO₃ type) percolates the earth's surface and starts reacting with minerals present within the aquifer system. In Fig. 4b, the bivariate plot between total cation and alkali earth minerals (Ca+Mg) illustrates that the groundwater is rich in Ca-Mg. The average ionic concentration of Na is 53.05 mg/L in arsenic-unaffected groundwater, which is more than that of Ca (the average value is 47 mg/L). On the other side, the arsenic-affected groundwater shows a higher average concentration of Ca (46 mg/L)

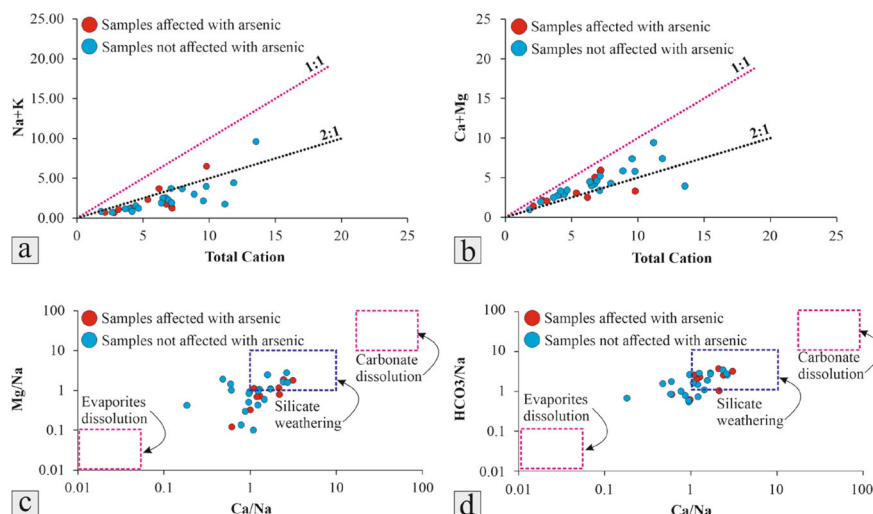


Fig. 4 Bivariate plots showing (a) Na^+K^+ vs. total cation; (b) $\text{Ca}^{2+}\text{Mg}^{2+}$ vs. total cation; (c) ionic ratio plot of $\text{Mg}^{2+}\text{-Na}^+$ vs. $\text{Ca}^{2+}\text{-Na}^+$, and (d) ionic ratio plot of $\text{HCO}_3^-\text{-Na}^+$ vs. $\text{Ca}^{2+}\text{-Na}^+$.

than Na (41.5 mg/L). This may result from the weathering of silicate minerals or the dissolution of carbonate minerals. However, the ion exchange process may also lead to adding and replacing some ionic constituents during the geochemical evolution along the groundwater flow path direction.

The bivariate plots, such as Ca-Na versus Mg-Na and Ca-Na versus $\text{HCO}_3\text{-Na}$, can be explained to interpret the controlling geochemical process that causes ionic variation in groundwater. In Fig. 4c, the scatter plots of some groundwater samples show that $\text{Ca-Na} > 1$ and $\text{Mg-Na} > 1$, which are due to the weathering of ferromagnesian silicate minerals (Pradhan et al., 2022). However, some groundwater samples are partially affected by the evaporation process, where $\text{Ca-Na} < 1$ and $\text{Mg-Na} < 1$. Similarly, groundwater with $\text{Ca-Na} > 1$ and $\text{HCO}_3\text{-Na} > 1$ is caused by the chemical weathering of silicate minerals as shown in Fig. 4d.

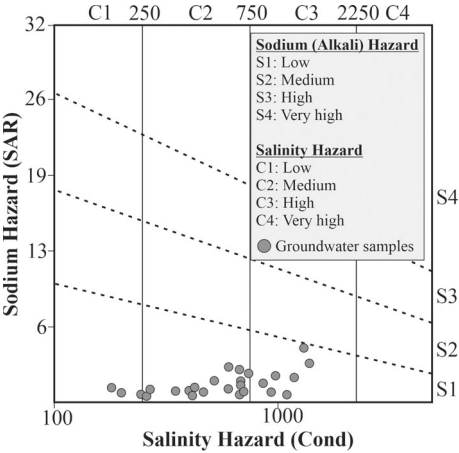
4.4 Drinking and Irrigation Water Quality

Davis and DeWiest (1966) categorized groundwater according to its content of total dissolved solids (TDS), with 500 mg/L (considered suitable for drinking), 500-1,000 mg/L (acceptable for drinking), and up to 3,000 mg/L (suitable for agricultural purposes). The US Salinity Laboratory (1954) categorized groundwater on the basis of EC (Table 3). The water quality data has been plotted on the Wilcox diagram (Fig. 5) to establish the irrigation quality of groundwater samples. In irrigation quality classification, the groundwater samples are in the C1S1, C2S1 and C3S1 classes.

Table 3 Water quality classification (US Salinity Laboratory, 1954)

<i>SAR (meq/L)</i>	<i>EC (μS/cm)</i>	<i>Water class</i>
Up to 10	Up to 250	Excellent
10-18	250-750	Good
18-26	750-2,250	Fair
>26	>2,250	Poor

Fig. 5 US Salinity Laboratory (1954) graph showing the suitability of water for irrigation usage.



Because EC is below 1,500 $\mu\text{S}/\text{cm}$, there is no sodium hazard in the groundwater, and therefore the groundwater in this area is safe for irrigation.

4.5 Percentage of Sodium (%Na)

The percentage of sodium remains a significant element in examining sodium-related risks, typically expressed as percentage of sodium (%Na). Based on %Na and EC, Wilcox (1948) suggested a technique for the usage of irrigation water. The %Na can be obtained by Equation (1):

$$\%Na = \frac{Na + K}{Ca + Mg + Na + K} \times 100, \tag{1}$$

where all the measurements of ion concentrations are expressed in terms of meq/L. Using the above equation, the classification of water is based on %Na as follows: three groundwater samples are in the water class ‘excellent’ (<20%), 19 samples are in the water class ‘good’ (20-40%), six samples are in the ‘permissible’ (40-60%) range and two samples belong to water class ‘doubtful’ (60-80%).

4.6 Sodium Adsorption Ratio (SAR)

The SAR is used to measure sodium (alkali) hazard to the soil. The assessment of sodium or alkali hazard in water used for irrigation is expressed through the computation of SAR using Equation (2):

$$\text{SAR} = \frac{\text{Na}}{\sqrt{(\text{Ca} + \text{Mg})/2}} \times 100, \quad (2)$$

Using the above equation, SAR values ranged from 0.7 to 4.66 meq/L with an average of 1.54 meq/L. Based on the classification of the US Salinity Laboratory (1954), all the samples are in the excellent category for irrigation purposes.

4.7 Relation of Arsenic with Other Ions

The groundwater composition in more than five locations, where arsenic levels are found in a variety of rock types, which are as follows; rhyolite (Joratarai [A-5] and Kaurikasa [A-29]), rhyolite tuff (Jadutola [A-6] and Muleitola [A-9]), and granite (Sonsaytola [A-23]). An attempt is made to correlate arsenic with other major and minor ions existing in the groundwater. The findings indicate that elevated arsenic concentrations are linked to the samples exhibiting low EC, reduced bicarbonate levels, and relatively higher sulphate content.

5 Conclusions

Groundwater in the arsenic-affected Ambagarh and its suburbs is assessed to understand the hydrogeochemical properties of groundwater resources for domestic and irrigation purposes. The geochemical analysis indicates that the groundwater in the region is typically fresh, displaying EC levels of 182–1,388 $\mu\text{S}/\text{cm}$, with only four wells showing values exceeding 1,000 $\mu\text{S}/\text{cm}$. Piper trilinear plot of groundwater samples illustrates that the Ca and Mg ions are dominant over Na and K ions. Two major dominant hydrogeochemical facies (Ca-HCO_3 and Ca-Cl-HCO_3) are identified. In addition, the groundwater displays an alkaline quality, with pH levels ranging between 7.6 and 8.2. In the Gibbs plot, the maximum groundwater samples are confined in the rock–water interaction and evaporation dominance area, suggesting that evaporation dominance and weathering of rock-forming minerals influence quality of the water. Rock–water interaction regions indicate that the major source of mineral concentration of dissolved ions is derived from the host rock whereas evaporation dominance regions suggested high temperature prevailing in the study area that resulted from evaporation proportion. However, the ion exchange process through

the weathering of silicate minerals is the controlling mechanism for the derivation of major chemical changes in aquifer systems. Moreover, the analysis of water samples using different chemical indicators, such as %Na and SAR, revealed that groundwater in the study area was in the good to excellent range for irrigation purposes. The US salinity diagrams of all groundwater samples were in the C1S1, C2S1, and C3S1 classes. Because of low values of EC and no sodium hazard, groundwater in the area was safe for irrigation use. It is observed that most of the groundwater in the study area is safe, except at few areas, viz. Kaurikasa (A-29), Joratarai (A-5), Sonsaytola (A-23), Jadutola (A-6) and Muleitola (A-9) are close to the shear zone, which is high arsenic-affected; hence, proper precautionary measures must be taken.

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Chapter 10

Gold Mine Tailings Treatments: Biotechnological Approaches



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1 Introduction

Cyanide leaching is the most common process for gold extraction through selective gold dissolution by cyanide solutions (cyanidation process) (Verbrugge et al., 2021; Brüger et al., 2018). Approximately 90% of gold extracted from ore worldwide uses cyanidation (Park et al., 2017; Adams, 2016) due to the Minamata Convention on Mercury adopted by 137 countries around the world to significantly reduce mercury emissions, eliminating its use from artisanal and small-scale gold mining (UNEP, 2019). Since its toxicity, cyanide wastes need treatment. Several approaches have been proposed, such as physicochemical processes. Nevertheless, biotechnological approaches are still a more suitable alternative due to sustainability. For setting up a biotechnological cyanide waste treatment at a mine, several factors need to be studied, such as the selection of proper cyanotrophic organisms and bioprocess strategy, besides the mine-tailing characterization to identify specific pollutants that need to be biodegraded or assimilated. Thus, this text aims to point out the factors to consider for a suitable biotechnological treatment of goldmine tailings.

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2 Goldmine Tailings: Cyanide's Impact on the Environment and Human Health

It is estimated that 20% of total cyanide production worldwide is used in mining (Guo et al., 2014), and global consumption of cyanide is appraised at 1.5 million metric tons per year (Park et al., 2017). As a result, cyanide-bearing wastes are produced (cyanidation wastes). These wastes can be leaching residues, tailings, and spent process waters. Gold extraction is a two-stage process: the extraction stage with NaCN and lime dissolved with water, producing a barren cyanide solution (typically 100–500 ppm of NaCN). It can be done by heap leach (sprayed with alkaline NaCN solution and a percolate down through the ore, resulting in a solution captured into a holding pond for further processes) or vat/tank leach (crushed ore is leached into large and closed tanks where gold is dissolved and adhered to carbon). Both methods use the highly soluble gold complexes with cyanide according to the reaction in Fig. 1.

$\text{NaAu}(\text{CN})_2$ stable ions formation brings gold into the solution and is recovered by carbon adsorption (Razanamahandry et al., 2017). As free cyanide is needed to dissolve gold, pH needs to be kept >9.2 to promote the formation of CN^- ion. Since ores not only have gold but also have metalloids exceeding precious metal concentrations by several magnitude orders, cyanide ions may also dissolve As, Cd, Co, Cu, Hg, Mo, Ni, Fe, Pb, Zn (Kandasamy et al., 2015). Another metal dissolution depends on its presence in the ore: completely weathered and oxidized ores have

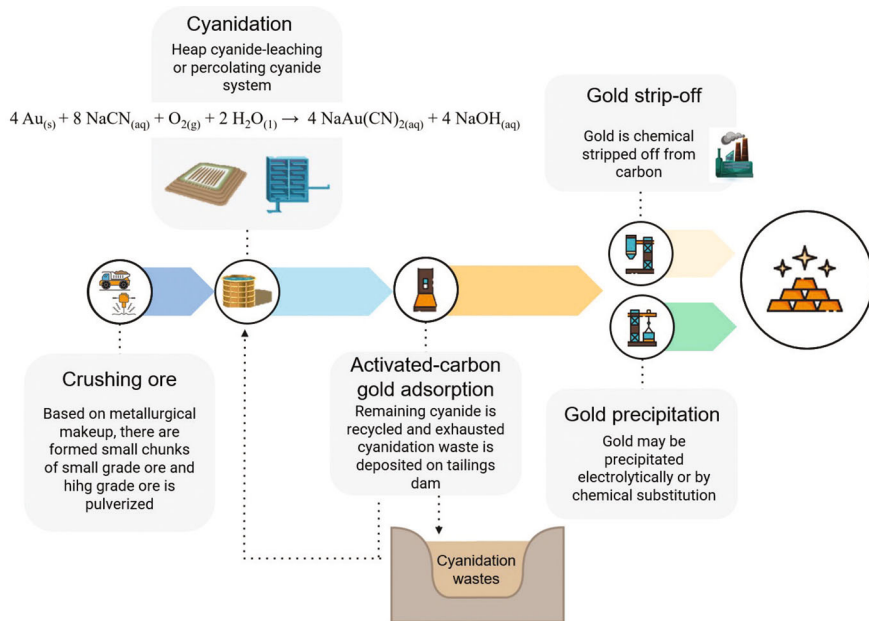


Fig. 1 Gold extraction by cyanidation process

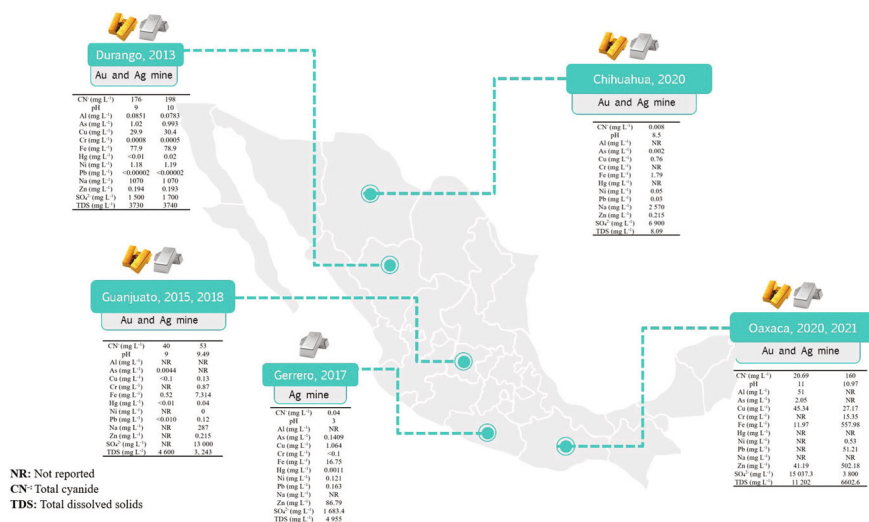


Fig. 2 Cyanided mine-tailings record in Mexico

metalloids as secondary minerals; sulfide-rich ores may be oxidized and react with cyanide, forming soluble thiocyanate and metal cyanides complexes. In our research group, we have documented the mine-tailings history presented in Fig. 2 for different Mexican mines in which different compositions can be noticed depending on the ore's nature.

Artisanal gold mining compresses mercury amalgamation and the consequent pollution with mercury and cyanide (Razanamahandry et al., 2017), which should be stopped since the Minamata Convention on Mercury was signed. Cyanides are a group of toxic compounds with the cyano ion ($\text{C}\equiv\text{N}^-$) (Kuyucak & Akcil, 2013), classified into five groups: free (F-CN^-), simple, weakly complexed, moderately strong complexed, and strongly complexed cyanides. Weak acid dissociable (WAD) are F-CN^- , weak and moderately strong complexed cyanides that, when the pH is lowered to <9.2 can be released as HCN, which is reasonably volatile and can be dispersed into atmosphere. Their chemical properties, nomenclature, and examples are presented in Fig. 3, in which their toxic effect on fish can be noticed.

Cyanides, particularly F-CN^- , are highly toxic due to their strong capacity to bind with metals. Therefore, the more stable the complex, the less harmful it is (Jaszczak et al., 2017). This carries out the inactivation of metalloenzymes such as cytochrome c oxidase, thereby inhibiting aerobic respiration (Mahendran et al., 2019; Luque-Almagro et al., 2016). In humans, once cyanide is absorbed into the bloodstream, it equilibrates CN^- and HCN. The latter can enter the cell's membranes and inhibit multiple enzymes, such as succinic dehydrogenase, superoxide dismutase, and cytochrome c oxidase (Mahendran et al., 2019). For mammals, cyanide's lethal dose is $0.5\text{--}3.5 \text{ mg kg}^{-1}$ (Jaszczak et al., 2017; Kumar et al., 2016); for humans, it

Free cyanide						
Common name	Example	IUPAC name	Stability constant*	Relative toxicity	Fish toxicity (mg L ⁻¹)	
WAD	CN ⁻ , HCN _{aq}	Cyanide ion / Hydrocyanic acid	NA	High	20-640	Chronic effects: 5-20
Simple compounds						
Common name	Example	IUPAC name	Stability constant*	Relative toxicity	Fish toxicity (mg L ⁻¹)	
WAD	NaCN _(s)	Sodium cyanide	2.24(10) ⁷	Medium to high depending	0.02-0.3	
	KCN _(s)	Potassium cyanide	2.24(10) ⁷		0.02-0.3	
	Ca(CN) _{2(s)}	Calcium dicyanide	NA		39 (in rats)	
	Hg(CN) _{2(s)}	Mercuric cyanide	NA		20 (in rats)	
	Zn(CN) _{2(s)}	Zinc dicyanide	11.1		0.02-0.3	
	Cd(CN) _{2(s)}	Cadmium dicyanide	5.6		0.02-0.3	
	Cu(CN) _{2(s)}	Copper cyanide	NA		0.4-4	
	Ni(CN) _{2(s)}	Nickel dicyanide	NA		0.4 (pH 6.5) 730 (pH 8)	
	Ag(CN) _{2(s)}	Silver cyanide	NA		123 (in rats)	
Weak complexes						
Common name	Example	IUPAC name	Stability constant*	Relative toxicity	Fish toxicity (mg L ⁻¹)	
WAD	[Zn(CN) ₂] ²⁻	Tetracyanozincate (II)	19.6	Medium to high depending	NA	
	[Cd(CN) ₂] ²⁻	Dicyanocadmiate (II)	NA		16 (in rats)	
	[Ag(CN) ₂] ⁻	Tetracyanoargentate (I)	17.9		NA	
Moderately strong complexes						
Common name	Example	IUPAC name	Stability constant*	Relative toxicity	Fish toxicity (mg L ⁻¹)	
WAD	[Cu(CN) ₂] ⁻	Copper dicyanide	18.8	Medium to high depending	NA	
	[Co(CN) ₂] ⁻	Tetracyanocobaltate (II)	23.5		NA	
	[Ni(CN) ₂] ⁻	Tetracyanonickelate (II)	30.3	on environmental conditions	NA	
	[Ag(CN) ₂] ⁻	Dicyanoargentate (I)	11.8		NA	
	[Au(CN) ₂] ⁻	Dicyanoaurate (I)	20.5		NA	
Strong complexes						
Common name	Example	IUPAC name	Stability constant*	Relative toxicity	Fish toxicity (mg L ⁻¹)	
SAD	[Au(CN) ₂] ⁻	Dicyanoaurate (I)	27		NA	
	[Co(CN) ₂] ⁻	Hexacyanocobaltate (II)	64	Low	24-200	
	[Fe(CN) ₂] ⁻	Hexacyanoferrate (II)	35.4		300 (in darkness)	
	[Fe(CN) ₂] ⁻	Hexacyanoferrate (II)	43.6		<0.2 (in daylight)	
Thiocyanate and cyanate						
Common name	Example	IUPAC name	Stability constant*	Relative toxicity	Fish toxicity (mg L ⁻¹)	
Other	SCN ⁻	Thiocyanate ion	NA	Low	13-82	
cyanide species	CN ⁻ **	Cyanide ion	NA	High	25 (in rats)	
	CNCI ⁺ **	Carbonyl chloride	NA	High	11-19 µg L ⁻¹	

*log K at 25 °C

**The presence of these compounds is not clearly understood in cyanided wastes.

NA: not available information

Fig. 3 Cyanides

depends on the exposure way: from inhalation range 200–314 mg HCN m⁻³, ingestion 0.56–1.52 mg CN kg⁻¹ and dermal contact 100 mg CN kg⁻¹ (Razanamahandry et al., 2017). The cyanide toxicity threshold is estimated at 200 ppm for aerobic microorganisms and 2 ppm for anaerobes (Kuyucak & Akcil, 2013).

The main risk of cyanidation is related to the eventual failure of cyanided waste deposits and the HCN volatilization surrounding cyanidation facilities (Brüger et al., 2018). Volatilization can be minimized by maintaining low saline and alkaline conditions within the cyanided waters. It is a potential danger since cyanide can pollute water and be lethal if fish, animals, and humans take enough. There is also a high risk of spillages, transportation accidents, bursting of pipes containing cyanide solutions, overtopping dams, and tailings dam failures. Cyanidation wastes such as process water and tailings must be treated to reduce their environmental and human health impact. Once precious metals are recovered, process water may contain aqueous F-CN⁻, cyanide metal complexes, cyanates, thiocyanates, and other chemical species from the ore. Most of this water is recovered and recirculated during cyanidation; however, this solution, with an excessive concentration of thiocyanates and metal cyanide complexes, is discarded (due to its exhausted extraction capacity). In the best scenario, exhausted cyanided water may be treated to reduce cyanide levels before discharge. Thus, it is pumped to tailing dams or discharged into water bodies and cyanided leached ore may be pumped as slurry to tailing dams. It is estimated that about one million tons of cyanided waste are produced worldwide each year and deposited in tailing dams (Roldán et al., 2021). Tailing dam's cyanided wastes are exposed to the ecosystem, so cyanide transport is possible. A cyanide cycle (Fig. 4) of complexed and F-CN⁻ involving physical, chemical, and biological reactions may

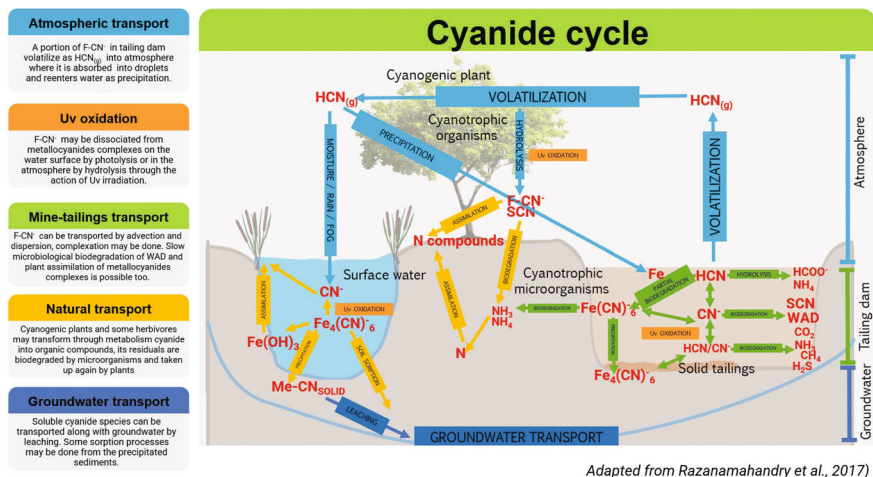


Fig. 4 Cyanide cycle

occur. In tailings, surface water complexation, precipitation, photolysis, volatilization, and biological oxidation may happen; biodegradation of WADs is possible in sediments. Lixiviation from underground soil to surrounding water bodies, uptake assimilation of complexed and F-CN^- by rhizosphere and cyanogenic plants may occur.

In goldmines, environmental impacts are not limited to cyanide's presence: mine tailings commonly have heavy metal and metalloid complexes, particularly Hg, As, Sb, and thiocyanates. Since cyanide complexes and metals can persist in waste repositories, safe levels must be achieved. Loss of wildlife and human health occurred at operating and rehabilitated mines where F-CN^- exceeded safe levels. Nations have implemented criteria for cyanide levels in food, drinking water, air, soil, and mining. For instance, the World Health Organization (WHO) sets a maximum permissible limit (MPL) of $70 \mu\text{g L}^{-1}$ for drinking water; the Environmental Protection Agency from the USA (USA-EPA) established MPL for free cyanide at $50 \mu\text{g L}^{-1}$ for freshwater bodies for aquatic life and $200 \mu\text{g L}^{-1}$ for drinking water. Compliance below-allowed limits can be achieved through cyanide treatment.

3 Goldmine Tailings Treatments

Different processes exist to convert cyanided wastes into less toxic compounds, especially the destruction of cyanide in tailings: natural, physicochemical, and biotechnological processes. Natural attenuation depends on local climatic and environmental conditions (Lottermoser, 2007). It includes volatilization, precipitation, adsorption, degradation by native bacteria or through UV radiation, natural oxidation to cyanate,

	 Methods: mechanisms involved	 Advantages	 Disadvantages
Natural	Attenuation: Volatilization, precipitation, or adsorption Oxidation: Through UV radiation Degradation: Biodegradation by indigenous microorganisms	No cost-related	Very slow rates and poor efficiency Only occurs on the surface
Chemical addition	Oxidation processes (SO ₂ -Air, Alkaline chlorination, Hydrogen peroxide, Ozonation, Caro's acid, UV catalyst); conversion to cyanate by hydrolysis. Iron sulfide process: Complexation Thermal treatment under high pressure: hydrolysis Fe/Cu precipitation: precipitation Cyanide recovery (SART, AVR, Cyanisorb): acidification, volatilization and condensation and reutilization, respectively.	Faster degradation rates. Effective at treating slurries. Well established. Simple to operate. High efficiencies. Recovery strategies make them very viable.	License payments. Adds hazardous expensive reagents and toxic intermediates. Excess precipitate accumulation SAD, ammonia, SCN, and chlorides are difficult to remove. Reacts preferentially with SCN. Requires removal of catalyst. Sludge accumulation problems.
Physical removal	Absorption processes (Ion Exchange, Activated Carbon Ion Flootation); complexation. Cyanide reduction: Dilution and volatilization, and slight microbial degradation and complexation.	Efficient and fast removal. Leads to CN recovery	Engineering costs: pumping capacity, installation, instrumentation, irrigation, and equipment residence time.

Adapted from: Kuyucak and Akcil, 2013; Mekuto et al., 2016

Fig. 5 Traditional treatment strategies

formation of thiocyanate, and complexation (chelation). Volatilization is supposed to be the primary mechanism of natural attenuation of tailing dams, but it was recently reported that this does not necessarily occur (Brüger et al., 2018).

Chemical methods compress the addition of substances that react with cyanide, forming fewer toxic compounds through oxidation (Razanamahandry et al., 2017) or a stabilization process. Some of those are alkaline chlorination, ozonation, SO₂/air (INCO), iron/copper precipitation, hydrogen peroxide oxidation, iron sulfide and sulfide precipitation, Caro’s acid and gas-filled membrane absorption (GFMA), AVR, SART, acidification, activated carbon polishing, ion exchange and reverse osmosis (Kuyucak & Akcil, 2013; Mekuto et al., 2016; Razanamahandry et al., 2017). Physical treatments are based on diluting cyanided wastewater without chemical reagents such as barren/freshwater rinse (Mekuto et al., 2016). Those methods are described in Fig. 5, in which the mechanisms involved and the technical advantages and disadvantages are presented. Biotechnological treatments are considered an attractive alternative due to their relatively low cost (compared to traditional methods) and because they can result in fewer hazardous by-products (Alvarado-López et al., 2023).

4 Advantages and Challenges of Physicochemical Treatments

Physicochemical methods for cyanided waste treatment relied on using hazardous reagents (Sharma et al., 2019) for stabilization and chemical oxidation of cyanide. They have been valuable and efficient for cyanide removal, but the chemical composition of cyanided waste, its volume, the effluent quality, and reagents availability

affect its feasibility (Sharma et al., 2019; Park et al., 2017). Some technical difficulties can be noticed in Fig. 5; for instance, alkaline chlorination is ineffective for some SAD due to slow reaction rates. This process produces sludge, requiring disposal licenses (Razanamahandry et al., 2017; Kumar et al., 2016). Another challenge is the cost-related issues: the addition of expensive compounds, pumping capacity, equipment installation, instrumentation, irrigation, and equipment residence time (Mekuto et al., 2016). Furthermore, the addition of compounds increases the total solids content of wastewater, making it undesirable for reuse and sometimes toxic to aquatic life (Razanamahandry et al., 2017). According to Shin et al. (2020), alkaline chlorination produces remaining chloramines and chlorine that cause secondary contaminations. However, these processes still do not solve the problem of secondary contaminations, including undesirable sulfate levels in the treated solution and the additional processes required to treat iron cyanide, ammonia and nitrate.

5 Biotechnological Treatments of Goldmine Tailings

Despite cyanided waste toxicity, some organisms are capable of detoxifying them. Goldmine tailings biotechnological treatments involve applying living organisms to remove cyanides from wastewater. Mine-tailings biotreatment uses organisms capable of changing complexed and F-CN⁻ to bicarbonate and ammonia, while metals are either adsorbed by biofilm or unconfined as precipitates in solution (Ibrahim et al., 2015). These treatments are considered the most eco-friendly, cheapest, efficient, cost-effective, and sustainable alternative for cyanided waste clean-up (Alvarado-López et al., 2023; Roldán et al., 2021; Cabello et al., 2018; Razanamahandry et al., 2017; Kumar et al., 2016; Ibrahim et al., 2015), besides its high capital cost (Ibrahim et al., 2015). Plants, bacteria, algae, and fungi use cyanided wastes as carbon or nitrogen sources for metabolism and are called cyanotrophic organisms. Fungi genera such as *Fusarium*, *Penicillium*, *Gloeocercospora*, and *Trichoderma*, algae such as *Scenedesmus obliquus* or *Chlorella vulgaris*, and bacteria like *Pseudomonas*, *Bacillus*, *Alcaligenes*, *Acinetobacter*, *Burkholderia*, *Rhodococcus*, *Klebsiella*, *Corynebacterium*, *Brevibacterium*, *Ralstonia* and *Serratia* have been reported as cyanotrophic (Sharma et al., 2019; Razanamahandry et al., 2017; Kumar et al., 2016; Ibrahim et al., 2015). Along with metabolic pathways for cyanide biodegradation or assimilation, cyanotrophic organisms should have mechanisms to prevent cyanide poisoning, like alternative oxidases (Roldán et al., 2021). Alkaline conditions should be maintained for cyanided waste treatment to avoid volatilization of F-CN⁻; thus, cyanotrophic used for this purpose should be alkaliphilic (organisms whose optimal pH occurs at pH 9–12).

Biotreatment of cyanide may be performed in two steps: the biooxidation of cyanide to cyanate using cyanide as C and N source, and its hydrolysis to ammonium and carbonates (Alvillo-Rivera et al., 2021; Razanamahandry et al., 2017). This could be done by different metabolic pathways that will be described later. Under aerobic conditions, cyanotrophics break down HCN and produce cyanate that undergoes

hydrolysis into ammonia and carbon dioxide. Without oxygen, anaerobic biodegradation can only happen in the presence of HS_2 or H_2S , which depends on the pH. At $\text{pH} \geq 7$, HS_2 dominates, and at $\text{pH} \leq 7$, H_2S dominates. This process may generate biogas, but anaerobic biodegradation is slower than aerobic and more vulnerable to cyanide toxicity (Ibrahim et al., 2015). More than one pathway can be used to biodegrade cyanide depending on environmental factors besides the genetic and physiological characteristics of the cyanotrophic organism (Alvarado-López et al., 2023; Ibrahim et al., 2015). Since goldmine tailings are formed by metalocyanide complexes from goldmine tailings, its bioassimilation is modified as metal cyanides and are oxidized by bioabsorption or precipitation. Degradation of cyanided metal complexes decreases in order $\text{Zn} > \text{Ni} > \text{Cu} > \text{Fe}$, according to Alvillo-Rivera et al., (2021) and Kumar et al., (2016). Sharma et al. (2019) reported that temperature and pH are essential for cyanide biodegradation since cyanide-degrading enzymes are produced by mesophilic cyanotrophics (Kumar et al., 2016). For bacteria, optimal temperatures range from 25–50° C, 43° C for fungi, and 40° C for plants. Optimal pH operating conditions are set from 5.2 to 10.5 for bacteria, 6 to 8.5 for fungi, and 12 for plants. The thiocyanate biodegradation, pH must be 10 (Kumar et al., 2016). Thus, for cyanide wastes, biodegradation needs to be fulfilled by alkaliphilic cyanotrophic organisms.

6 Cyanides Biodegradation Metabolism

Different enzymatic pathways are related to cyanide's biotransformation and biodegradation processes. Degradative reactions are hydrolytic, oxidative, and reductive pathways. Assimilative routes are substitution, transfer, and synthesis reactions (Gupta et al., 2010). Degradative pathways lead to the transformation of cyanide into simple molecules, and the assimilative utilize cyanide as a carbon or nitrogen source for the synthesis of metabolites (Mahendran et al., 2019). Although most cyanotrophic are aerobes, anaerobic biodegradation of cyanide may be possible by reductive or hydrolytic pathways. In Fig. 6, different cyanide biotransformation pathways are presented among its mechanisms.

Hydrolytic reactions may be catalyzed by cyanide hydratase (CHT) (fungi) yielding formamide and by cyanidase (CynD) (bacteria) forming formate and ammonia when cyanide reacts with water. Both enzymes have molecular resemblances to nitrilases and nitrile hydratases, which are useful for the hydrolytic degradation of organic cyanides (nitriles), which are not commonly present in mine tailings. CHT has exceptional substrate specificity but requires an extra carbon source. CynD, on the other hand, increases its activity when metal ions are present, and it does not require cofactors (Gupta et al., 2010).

Oxidative reactions compress the oxygenolytic formation of cyanate by cyanide monooxygenase (CMO) or cyanide dioxygenase (CDO). Then, it changes to ammonia and CO_2 with NADPH and an extra carbon source, depending on cyanases (Cys) (Gupta et al., 2010). Cys were identified in bacteria, plants, and animals. Its role

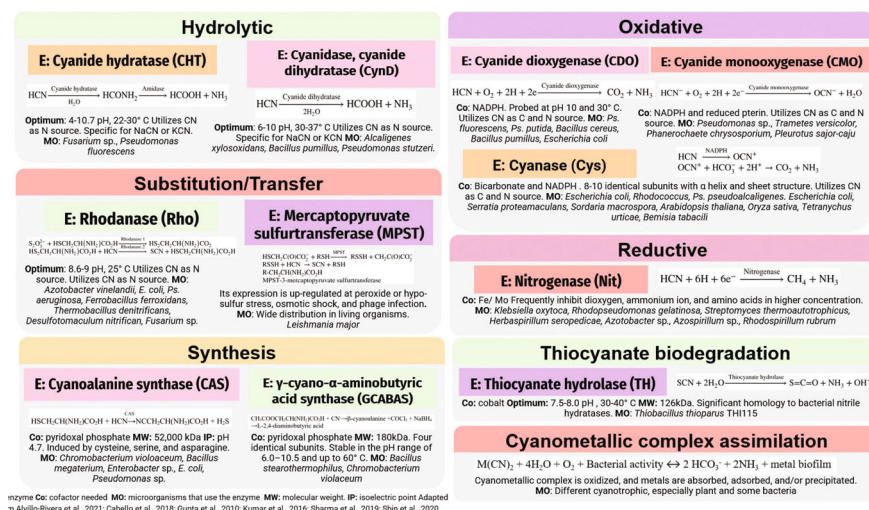


Fig. 6 Cyanide biodegradation pathways

is related to defense against poisoning and to nitrogen/bicarbonate/CO₂ metabolism, or in plants, for ammonia absorption, concentration, and deliverance of CO₂ for photosynthesis (Ibrahim et al., 2015). CDO, conversely, is related to the production of cyanohydrin complexes for cyanide biodegradation (Kumar et al., 2016).

Reductive reactions compress the transfer of an electron pair to cyanide, forming ammonia and methane with the action of nitrogenase (nit), denitrifying cyanides to gaseous nitrogen under anaerobic conditions (Kumar et al., 2016; Gupta et al., 2010). Nit breaks down cyanide's triple bond, getting electron donors and bonding nitrogen to hydrogen atoms in individual reactions for each bond, requiring a MoFe apoprotein and ATP following a great deal of chemical energy (Gupta et al., 2010). Anaerobic bioreactors favor methanogenic organisms and biogas production, but high cyanide concentration in cyanided wastes inhibits methanogenesis of biogranules (Ibrahim et al., 2015). Bacteria such as *Klebsiella oxytoca* have shown this ability (Kumar et al., 2016).

Synthesis pathways are common in bacteria that use cyanoalanine synthase (CAS) as a catalyst to convert acetylserine and cyanide into β -cyanoalanine (Ibrahim et al., 2015), preventing cyanotrophic cells from cyanide toxicity and providing an extra nitrogen source for bacterial growth (Kumar et al., 2016). Synthesis of γ -cyano- α -aminobutyric acid by γ -cyano- α -aminobutyric acid synthase (GCABAS) also falls into this pathway group (Park et al., 2017). GCABAS requires pyridoxal phosphate and is induced by amino acids like glutamate and glycine (Gupta et al., 2010).

Substitution/transfer pathways involves cyanide assimilation through the action of rhodanase (Rho) and mercaptopyruvate sulfurtransferase (MPST) tending to increase cyanotrophic growth rate, due to the extra nitrogen source and preventing cyanide toxicity (Gupta et al., 2010). Rhodanase are highly conserved widespread

enzymes that transfer sulfur groups from a suitable donor like thiosulfate to cyanide, compressing the formation of sulfite and thiocyanate (Gupta et al., 2010). MPST converts 3-mercaptopyruvate into thiosulfate in a two-step mechanism, the second step intermediate reacts with cyanide forming pyruvate and thiocyanate (Gupta et al., 2010).

In mine tailings from sulfide-rich ores, sulfurtransferase (ST) conducts its biodegradation. There are at least two different pathways for this purpose: cyanate pathway and carbonyl pathway, in which thiocyanate is transformed into ammonia and carbonyl sulfide with the action of thiocyanate hydrolase (TH) in chemolithotrophic organisms (Ibrahim et al., 2015).

Sometimes, a cyanotrophic may utilize more than one pathway for cyanide biodegradation depending on factors such as the mechanism of cyanide tolerance (alternative oxidases for electron transport chain (AOX), adsorbent biofilm formation, or siderophores production) and the way cyanide-metal complex dissociation or chelating metals (Gupta et al., 2010).

7 Cyanotrophic Organisms

7.1 Plants

Some plants are cyanotrophic since they can remove cyanide from soil and water. Plant-microbe interactions from the rhizosphere emit secondary metabolites and exude biosurfactants (rhamnolipids), helping to increase the solubility and bioavailability of organic cyanides (Sharma et al., 2019; Kumar et al., 2016). Siderophores may also be present in roots, influencing sequestration and translocation of metallocyanide. Fungi engaged in symbiotic association with plants increase root surface area and detoxify cyanide through stabilization, extraction, and degradation by rhizoremediation (Kumar et al., 2016). Root uptake of cyanide is not easy: the suberine layer from plant root acts as an impermeable membrane resisting cyanide flow to xylem by its apoplast, governed by diffusion, depending on the hydrophobicity of cyanides. Once entered plant tissue, metallocyanides and nitriles are bound to chelators and actively transported to vacuoles. At vacuoles, conjugated compounds form complex structures that react with sulfide and can stay accumulated within the plant tissue or metabolize partially into intermediates by enzymes from endophytic microorganisms. Those enzymes add side groups that increase solubility and initiate degradation by phytoenzymes such as CAS, Rho, formamide hydrolase, or CynD. Phytoremediation is based on the use of plants for cyanide detoxification. Cyanogenic and not cyanogenic species like maize and Chinese vegetation were reported as cyanide degraders. Plants capable of treating cyanides from soils and wastewater are listed in Table 1, which have a low tolerance (<10 mg L⁻¹) and slow cyanide degradation rates, require huge land area, and are practically not feasible for arid regions (Sharma et al., 2019).

Table 1 Cyanotrophic plants

Specie	Biodegradation capacity (mg CN kg ⁻¹ h ⁻¹)	Conditions	References
<i>Sambucus chinensis</i> Lindl	8.77	Not reported	Kumar et al. (2016)
<i>Torilis japonica</i> (Houtt.) DC	7.52		
<i>Prunus persica</i> sleb	2.83		
<i>Prunus pseudocerasus</i> Lindl	6.28		
<i>Salix babylonica</i> L	6.08		
<i>Glycine max</i> (L.) merr	3.45		
<i>Chimonanthus praecox</i> (L.) Link	5.97		
<i>Liquidambar formosana</i> Hance	5.22		
<i>Metasequoia glyptostroboides</i> Hu & Cheng	5.32		
<i>Capsicum frutescens</i> L.cv. 'Hexiniujiangjiao'	4.93		
<i>Euonymus alatus</i> (thunb.) sieb	4.27		
<i>Zea mays</i> L	4.75		
<i>Cudrania tricuspidata</i> (Carr.) Bur	4.33		
<i>Sorghum vulgare</i> Pers	3.98		
<i>Calendula officinalis</i> L	4.02		
<i>Nymphaea teragona</i>	36		
<i>Salix matsudana alba</i>	3.15		
<i>Alternanthera philoeroides</i> Griseb	3.97		
<i>Populus deltoides</i> marsh	3.43		
<i>Iris tectorum</i> Maxim	3.05		
<i>Prunus persica</i> (L.) Batsch	2.83		
<i>Buxus sinica</i> (rehd. & Wils.) m. Cheng	2.72		
<i>Trachycarpus fortunei</i> (Hook.f.) H. Wendl	1.63		
<i>Viburnum odoratissimum</i> Ker-Gawl	2.28		
<i>Gossypium hirsutum</i> L	1.52		

(continued)

Table 1 (continued)

Specie	Biodegradation capacity (mg CN kg ⁻¹ h ⁻¹)	Conditions	References
<i>Pterocarya stenoptera</i> C.DC	1.75		
<i>Podocarpus macrophyllus</i> (thunb.) D.Don	0.6		
<i>Credrus deodara</i> (roxb.) Loud	0.23		
<i>Eichhornia crassipes</i>	35	Leaf cuttings	Ebel et al., (2007)
<i>Sambucus chinensis</i>	12.6	30° C	Yu et al., (2005)
<i>Salix babylonica</i>	9.72	32° C	Yu et al., (2005)
<i>Arabidopsis thaliana</i> (transgenic)	1.2 mM 2.5 mM	Growth medium as foliar spray	Kebeish and Al- Zoubi (2017)
<i>Citrus sinensis</i>	100 mg L ⁻¹	50 °C pH = 12	Santos et al. (2013a, b)

7.2 Algae

Two bioremediation studies are found with cyanotrophic algae (Dwivedi et al., 2011). For instance, *Scenedesmus obliquus* reported by Gurbuz et al. (2009) concluded that cyanide biodegradation with this species is affected by its initial concentration, the amount of inoculated biomass, and the degradation time; in the study by Kiruthika (2008), a consortium of *Chlorella vulgaris* with *Pseudomonas fluorescens* was used for this purpose.

7.3 Fungi

Some cyanotrophic fungi convert cyanide into formamide, CO₂, and ammonia through hydrolytic pathways with cyanide hydratase. Fungi genera like *Fusarium*, *Gloeocercospora*, *Aspergillus*, *Rhizobium*, *Trichoderma*, *Scytalidium*, and *Penicillium* have been reported as cyanide degraders (Mahendran et al., 2019; Razanamahandry et al., 2017). Fungi are commonly used for cyanided soil bioremediation at pH conditions ranging 4–8.5 and temperatures up to 43° C, as presented in Table 2. The most effective fungi reported for cyanide biotreatment was *Fusarium oxysporum* N-10, with a cyanide biodegradation rate of 1 mM d⁻¹ (Razanamahandry et al., 2017).

Table 2 Cyanotrophic fungi

Specie	Cyanide source	Carbon source	[CN ⁻] ₀ (mg L ⁻¹)	Culture conditions			Biodegradation capacity (%)
				T (°C)	pH	Agitation (rpm)	
<i>Trametes versicolor</i>	KCN	NaOH/NaHCO ₃	100	30	10.5	100	75
<i>Fusarium oxysporum</i> EKT01/02	KCN	Nutrient broth	100	25	10	140	77.6
<i>Aspergillus awamori</i>	KCN	Citrus waste (1%)	280	30	>9	NR	80
<i>Trichoderma koningii</i>	CN	None	500	28	6.5	140	90
<i>Fusarium oxysporum</i> CCM1 876	CN	Czapek stock solution	0.3	30	8	NR	86
<i>Fusarium solani</i>	KCN	NR	0.5–0.8*	30	9.2–10.7	NR	NR
Mixed culture of <i>F. solani</i> , <i>T. polysporum</i> , <i>F. oxysporum</i> , <i>S. thermophilum</i> , <i>P. miczynsky</i>	K ₂ Ni(CN) ₄ K ₄ Fe(CN) ₆	NR	0.75* 0.25–1*	25	4.5–7	NR	50–56
<i>Trichoderma</i> sp. and <i>Fusarium</i> sp.	CN	NR	2000	25	6.5	NR	>99
<i>Fusarium oxysporum</i>	CN and formamide	NR	1–7*	25–30	8	NR	96
<i>Fusarium solani</i> and <i>Trichoderma polysporum</i>	Iron or nickel cyanide	NR	0.5*	NR	NR	NR	50
<i>Fusarium solani</i> and <i>Trichoderma harzianum</i> , <i>Trichoderma pseudokoningii</i>	KCN	NR	0.125*	NR	NR	NR	
<i>Trametes versicolor</i>	Free cyanide	Citrate	400	30	10.5	NR	>99
<i>Gloeocercospora sorghi</i>	KCN metal cyanide complexes	NR	20*	27–43	6–8.5	NR	<80
<i>Fusarium oxysporum</i> and <i>Beta vulgaris</i>	Metal cyanide + KCN	NR	500	NR	NR	NR	83

[CN⁻]₀: initial cyanide concentration. NR: not reported information. *mM

Adapted from: Alvillo-Rivera et al. (2021), Ibrahim et al. (2015), Kumar et al. (2016), Mekuto et al. (2016), Razanamahandry et al. (2017), Akinpelu et al. (2018)

7.4 Bacteria

Bacteria-based biodegradation processes consist of various bacteria that play an individual metabolic role in each degradation step, so community dynamics are vital for controlling and operating a biotechnological treatment process (Shin et al., 2020). These dynamics are necessary to establish optimal operating conditions for cyanide's biodegradation in mining effluents. Table 3 shows that different operating conditions for cyanotrophic bacteria are reported for different cyanide sources. Bacteria genera like *Pseudomonas*, *Bacillus*, *Alcaligenes*, *Acinetobacter*, *Brevundimonas*, *Burkholderia*, *Rhodococcus*, *Klebsiella*, *Corynebacterium*, *Brevibacterium*, *Ralstonia*, *Thiobacillus*, *Halomonas*, *Neurospora*, *Gibberella*, *Micromonospora*, *Microbacterium*, *Citrobacter*, *Thioalkalimicrobium*, *Thioalkalivibrio*, *Enterobacter*, *Chroococcus*, *Azotobacter*, *Cellulosimicrobium*, *Providencia* and *Serratia* have been reported as cyanotrophic (Alvarado-López et al., 2023; Sharma et al., 2019; Razanamahandry et al., 2017; Kumar et al., 2016; Ibrahim et al., 2015). The best cyanide biodegradation capacity was shown by *Micromonospora braunna* (Shete & Kapdnis, 2012) (isolated from garden soil), which degrades 1000 mg L^{-1} of KCN in a dextrose-added medium, while in real mine-tailings without any extra nutrient source, the best cyanotrophic was *Halomonas* degrading 76% of 50 mg L^{-1} of total cyanide (Khamar et al., 2015).

Indigenous bacteria

Indigenous bacteria (isolated from the same medium that will be bioremediated) have advantages over bacterial collection strains due to adaptation to complex media with toxic or recalcitrant composition and environmental conditions (Ibrahim et al., 2015). Diversity and distribution of microorganisms are influenced by the nature of the geochemical matrix from microhabitats and the presence of toxic xenobiotics polluting compounds that cause changes in the structure, diversity, and function of the microbial community site-associated (López-Ramírez et al., 2021), making them suitable for biotreatments. For mine tailings, alkaline conditions, heavy metal presence, and cyanide toxicity have promoted changes in indigenous cyanotrophic that make them suitable for its treatment. Table 4 shows the reports of this type of bacteria that have been used to remove cyanide in alkaline media and cyanide wastes. It is noteworthy the higher biodegradation efficiency than other cyanotrophic and the probed cyanide biodegradation from *Bacillus* sp. (Guadalima and Monteros 2018) and *Halomonas* (Khamar et al., 2015), which have been tested on tailings. Bacterial strains 2DIII and 2AII (Dimitrova et al., 2020) that biodegraded iron cyanided complexes were also remarkable. The challenge for these bacteria is related to their isolation and difficulty in cultivation (99% are non-cultivable). Thus, culture-independent technologies must be carried out to study them (Roldán et al., 2021).

Table 3 Cyanotrophic bacteria

Bacterial specie	Cyanide source	[CN ⁻] ₀ (mg L ⁻¹)	Cultivation mode	Media	Operational conditions			Biodegradation efficiency (%)	References
					Agitation (rpm)	pH	T CQ		
Bacterial consortium	NaCN	120	SC	Modified M9	150	>9.6	30	>95	Alvarado-López et al. (2023)
<i>Bacillus</i> and <i>Enterococcus</i>	NaCN	200	SC	LB	180	10	30	21	López-Ramírez et al. (2021)
<i>Bacillus subtilis</i> BEB1	NaCN	130	SC	Nutrient broth	NR	7	28	53.8	Al-Badri et al. (2020)
Bacterial strains 2DIII and 2AII	K ₄ [Fe(CN) ₆]	50	SC	N-free MM, enriched with a Fe-free TES	160	7.3	NR	20	Dimitrova et al. (2020)
Not identified bacterial consortia	KCN	300	SC	Mining wastewater	120	8.5	25	>99	Razanamahandry et al. (2019)
<i>Pseudomonas putida</i> y <i>Pseudomonas stutzeri</i>	KCN	340	SC	Nutrient broth with phenol	160	7	28	80.6	Singh et al. (2018)
<i>Bacillus</i> sp.	CN	300	RBC	Mining wastewater, salts, and sucrose	NR	10	20–25	96.8	Guadalima and Monteros (2018)
<i>Pseudomonas mendocina</i> P6115	KCN	78	SC	M9	150	10.5	25	80.5	Miranda-Carrasco et al. (2018)
<i>Bacillus</i> sp. (CDO) y <i>Bacillus</i> , <i>Klebsiella</i> , <i>Providencia</i> , <i>Pseudomonas</i> (TDO)	SCN and CN	200	SC	K ₃ PO ₄ glucosa, cyanide	180	9.9	30	9989	Mekuto et al. (2018)

(continued)

Table 3 (continued)

Bacterial specie	Cyanide source	[CN ⁻] ₀ (mg L ⁻¹)	Cultivation mode	Media	Operational conditions			Biodegradation efficiency (%)	References
					Agitation (rpm)	pH	T CQ		
<i>Pseudomonas pseudoalcaligenes</i> CECT 5344	NaCN, Thiocyanate, metalocyanides complexes	30* (cyanide) 100* (cyanate)	SC	NR	NR	9	30	100	Cabello et al., (2018)
<i>Pseudomonas parafulva</i> NBRC 16,636(T)	KCN	200	SC	Sterile soil, K ₃ PO ₄ glucose and yeast extract	150	9.5	32	93.50	Moradkhani et al. (2018)
<i>Bacillus</i> sp M01PTCC 1908	KCN	260	SC	Nutrient broth	130	6.8	44	86	Javaheri Safa et al, (2017)
Bacterial consortium	KCN	80	SC	Synthetic medium, glucose	200	9.5	30	95	Park et al. (2017)
<i>Klebsiella pneumoniae</i> ATCC 13,883	KCN K ₄ Fe(CN) ₆ .3H ₂ O, Na ₄ Fe(CN) ₆	12.84 78 78	SC	Glucose	150	7	21	>99 85 87.5	Avcioglu and Bilkay (2016)
<i>Bacillus cereus</i>	KCN	100	SC	BM	150	7	35	99.63	Dwivedi et al., (2016)
<i>Pseudomonas pseudoalcaligenes</i> CECT 5344	Free cyanide from jewelry wastewaters	2*	5 L batch reactor	M9 with CH ₃ (CH ₂) ₆ COONa and jewelry waste	450	9.5	30	100	Manso Cobos et al. (2015)
<i>Pseudomonas</i> sp.	NaCN	655	SC	Salts, carbonate buffer	NR	9.5	36	66	Morillo Mendoza and Guevara Gonzales (2015)

(continued)

Table 3 (continued)

Bacterial specie	Cyanide source	[CN ⁻] ₀ (mg L ⁻¹)	Cultivation mode	Media	Operational conditions			Biodegradation efficiency (%)	References
					Agitation (rpm)	pH	T CQ		
<i>Bacillus pumilus</i> and <i>Pseudomonas putida</i>	NaCN	655	SC	MSM, cyanide, glucose	125	NR	30	63 61	Kandasamy et al. (2015)
<i>Halomonas</i>	Total cyanide	50	SC	Mine tailings	150	9.5	30	76	Khamar et al. (2015)
<i>Bacillus</i> sp. CN-22	KCN	200	SC	Ammonium salts, potassium phosphates, magnesium sulfate, sodium chloride, KCN	193	10.3	31	99,369	Wu et al. (2014)
Not identified	KCN	200	SC	BM, glucose	100–200	10.3	34.2	86	Minizadeh et al. (2014)
Consortium dominated by <i>Bacillus safensis</i> , <i>licheniformis</i> and <i>tequilensis</i>	KCN	400	SC	Agricultural wastewater	NR	9.5	37	60	Mekuto et al. (2013)
<i>Pseudomonas Fluorescens Pf-5</i>	K ₄ Fe(CN) ₆ ·3H ₂ O, Na ₄ Fe(CN) ₆ ·10H ₂ O y KAg(CN) ₂	975.48 973.65 324.01	SC	M9, glucose	230	7	30	53.8	Bouari et al. (2013)

(continued)

Table 3 (continued)

Bacterial specie	Cyanide source	[CN ⁻] ₀ (mg L ⁻¹)	Cultivation mode	Media	Operational conditions			Biodegradation efficiency (%)	References
					Agitation (rpm)	pH	TCQ		
Sulfate reducing bacteria isolated from mine tailings and cyanotrophic (BC) <i>Pseudomonas</i> sp.	CN ⁻ (from tailing)	585	Amerobic columns	Mine tailing, sodium lactate, acetate phosphate	Anareobic	9	20	88	Hurtado and Berastain, (2012)
<i>Micromonospora braunna</i>	KCN	1000	SC	Dextrose, potassium phosphate, trace elements	NR	NR	30	98.79	Shete and Kapdnis, (2012)

CDO: Cyanide degrading organisms **TDO:** Thiocyanate degrading organisms **NR:** not reported information **[CN⁻ ~ I₀]**: initial cyanide concentration **SC:** suspended cells **MSM:** Mineral minimum medium. *mM

Table 4 Indigenous cyanotrophic bacteria

Microorganism	Isolation site	Cyanide source	[CN ⁻] ₀ (mg L ⁻¹)	Biodegradation efficiency (%)	References
<i>Bacillus cereus</i> , <i>Microbacterium paraoxydans</i> , <i>Brevundimonas vesicularis</i> , <i>Cellulosimicrobium cellulans</i> , and <i>Brevibacterium casei</i>	Tailing pond (Mexico)	NaCN	120	>95	Alvarado-López et al. (2023)
<i>Bacillus</i> and <i>Enterococcus</i>	Cyanide gold-leaching plant (Colombia)	NaCN	200	21	López-Ramírez et al. (2021)
<i>Bacillus subtilis</i> BEB1	<i>Bougainvillea spectabilis</i>	NaCN	130	53.8	Al-Badri et al. (2020)
Bacterial strains 2DIII and 2AII	Contaminated soil	K ₄ [Fe(CN) ₆]	50	20	Dimitrova et al. (2020)
Not identified bacterial consortia	Wastewater	KCN	300	>99	Razanamahandry et al. (2019)
<i>Pseudomonas putida</i> and <i>Pseudomonas stutzeri</i>	Contaminated soil	KCN	300	80.6	Singh et al. (2018)
<i>Pseudomonas mendocina</i> P6115	Tailing pond (Mexico)	NaCN	78	80.5	Miranda-Carrasco et al. (2018)
<i>Pseudomonas parafulva</i> NBRC 16,636(T)	Goldmine soil	KCN	200	93.50	Moradkhani et al., (2018)
<i>Bacillus</i> sp.	Mine sediment (Ecuador)	CN ⁻ (from mine tailings)	300	96.89	Guadalima and Monteros (2018)
<i>Bacillus</i> sp. and <i>Bacillus</i> , <i>Klebsiella</i> , <i>Providencia</i> , <i>Pseudomonas</i>	Electroplating/gravimetric entrainment	SCN and CN	250	99 89	Mekuto et al. (2018)
<i>Bacillus</i> sp M01PTCC 1908	Mine wastewater / soil (Iran)	KCN	260	86	Javaheri Safa et al. (2017)

(continued)

Table 4 (continued)

Microorganism	Isolation site	Cyanide source	[CN ⁻] ₀ (mg L ⁻¹)	Biodegradation efficiency (%)	References
<i>Bacillus cereus</i>	Electroplating soil	KCN	100	99.63	Dwivedi et al. (2016)
Not identified	Burkina Faso basin	KCN	80	95	Razanamahandry et al. (2016)
<i>Halomonas</i> and a new taxon	Tailing pond (Iran)	Mine tailing	50	66	Khamar et al. (2015)
<i>Pseudomonas pseudocaligenes</i> CECT 5344	Guadalquivir River sludge	Free cyanide in jewelry wastewaters	2*	100	Manso Cobos et al. (2015)
<i>Pseudomonas</i> sp.	Activated sludge (Peru)	NacN	655	66	Morillo Mendoza and Guevara Gonzales (2015)
<i>Bacillus pumilus</i> and <i>Pseudomonas putida</i>	Wastewater	NacN	245	63 61	Kandasamy et al. (2015)
<i>Bacillus</i> sp.	Wastewater	HCN	700	NR	Wu et al. (2014)
Not identified	Wastewater	KCN	20	86	Mirzadeh et al. (2014)
Consortium dominated by <i>Bacillus safensis</i> , <i>licheniformis</i> and <i>tequilensis</i>	Electroplating	KCN	400	60	Mekuto et al. (2013)
<i>Micromonospora braunna</i>	Garden soil	KCN	1000	98.70	Shete and Kapdnis (2012)

* mM

7.5 Factors Affecting Cyanides Biodegradation

A successful biotreatment for cyanide wastes is ensured by the presence of cyanotrophic alkaliphilic microorganisms, which are affected by environmental factors while degrading cyanide: cyanide concentration, nutrients bioavailability, aeration, other contaminants presence, and its high concentration (Mahendran et al., 2019; Ibrahim et al., 2015; Kumar et al., 2016; Gupta et al., 2010).

The most essential cyanide biodegradation-affecting factor is the initial cyanide concentration. Few cyanotrophs can resist cyanide at initial concentrations higher than 500 mg L⁻¹ (Kumar et al., 2016). For instance, *Thioalkalimicrobium*, *Thioalkalivibrio* (Sorokin et al., 2001), *Klebsiella pneumonia* y *Ralstonia* sp. (Chaudhari & Kodam, 2010), *Micromonospora Brianna* (Shete & Kapdnis, 2012), or *Pseudomonas fluorescens* (Restrepo et al., 2006). Nutrient bioavailability is also essential: carbon source is the restrictive nutrient (Kumar et al., 2016). It is known that carbonyl groups from the typical carbon sources react with cyanides, forming carboxylic salt in the presence of metal ions, like in mine tailings, making carbon sources not responsible for microbial assimilation (Kumar et al., 2016). However, the cyanide biodegradation rate increased with glucose and had a promising effect on the degradation of metalocyanide complexes with *Klebsiella oxytoca* (Kumar et al., 2016). Nitrogen source is crucial: some bacteria prefer cyanide as a nitrogen source instead of low-molecular-weight proteins like *Pseudomonas*. Some fungi, such as *Fusarium solani*, can consume hexacyanoferrate as the sole nitrogen source, with glucose as an extra carbon source (Kumar et al., 2016). Regarding aeration, it is restrictive for oxidative pathways due to its requirement for the formation of cyanate, and it is important to recall that aerobic cyanide biodegradation is faster than anaerobic for most bacteria (Kumar et al., 2016). Other xenobiotic pollutants, such as heavy metals, may affect cyanide biodegradation by swaying the native populations and hindering the proliferation of specific organisms (Kumar et al., 2016; Ibrahim et al., 2015). Temperature and pH are the other important factors, as previously described.

For phytoremediation with cyanotrophic plants, it is crucial to consider the plant nature, pH and cation exchange capacity (CEC), bioaugmentation, and chelation in roots (Kumar et al., 2016). Firstly, plant species may be selected appropriately for suitable root depth and morphology; agronomic practices may be applied to maximize cyanide removal (plant root guidance through deep planting in a casing into cyanide-polluted water). The bioavailability of cyanide may be enhanced with chelators (siderophores, organic acids, or phenolics) releasing metal cations from cyanide complexes. Rhizosphere bacteria may release biosurfactants that make hydrophobic compounds as metalocyanides soluble and increase plant uptake (Kumar et al., 2016). pH and CEC control adsorption-desorption and precipitation-solubilization reactions. CEC depends on soil pH and ejecting protons through ATPase action, increasing metal-cations bioavailability. Cyanotrophic microorganisms' bioaugmentation in plant rhizosphere to degrade target pollutants may also favour cyanide phytodegradation.

7.6 Biotechnological Strategies

Phytoremediation is superior to other biotechnological strategies for metalocyanides (Fe or Hg) (Kumar et al., 2016; Sharma et al., 2019). Strategies like constructed wetlands are commonly designed for cyanide stabilization and biodegradation from solid and aqueous phases (Kumar et al., 2016). Cyanide treatment could be done with grasses, woody plants, or water hyacinths (Kumar et al., 2016). This strategy has been limited to laboratory studies with plants like *Eichhornia crassipes*, *Hordeum vulgare*, *Avena sativa*, *Sorghum bicolor*, *Salix babylonica*, *Sambucus chinensis*, *Torilis japonica*, *Linum usatassimum* and *Salix viminalis* (Mekuto et al., 2016). Maximum cyanide tolerance by plants has been reported at 50 mg L⁻¹ of total cyanide. Moreover, phytoremediation would require a vast land area and a dosage of cyanide administration (diluted at metabolizable concentrations < 10 mg L⁻¹ of F-CN⁻). Other disadvantages are related to using water limited in arid and semi-arid regions; additionally, minimal growth or nonexistent in different regions (Mekuto et al., 2016).

Microorganism cyanided waste treatment may be performed through suspended or immobilized biomass bioreactors. For the first strategy, cyanotrophics are completely mixed by mechanical meanings with different operational modes. Immobilized biomass systems are stationary or fixed beds where treated water moves through them. In a moving-bed biofilm reactor (MBBR), biomass growth increases biofilm on the support surface due to extracellular polymeric compounds and inside them due to depositing and biomass-trapping (Alvillo-Rivera et al., 2021). Biomass is immobilized on packing materials such as rocks, clay, or synthetic materials; for immobilized biomass systems, pH, temperature, and type of packing need to be considered for optimal design. Immobilization was reported effective and efficient on a laboratory scale due to higher specific surface for microbial proliferation, making cyanotrophic cells less vulnerable to toxic compounds and more tolerant to distress in the cyanided environment, eliminating the complex and expensive time-consuming recovery and biomass recycling (Ibrahim et al., 2015). For this strategy, supplying sufficient oxygenation is a significant challenge for keeping viable cells throughout the culture.

At the industrial level, biotechnological treatments were first used at Homestake Mine (South Dakota, USA) to reduce WAD below 0.2 mg L⁻¹ with an aerobic system of rotating biological contactor (RBC), degrading cyanide into ammonia followed by oxidation to nitrate in the second stage (Shin et al., 2020). Its main challenge is the eventual overflow of cyanide to secondary nitrifying and denitrifying stages, causing cyanide poisoning of noncyanotrophic bacteria (Mekuto et al., 2016). The same mining company implemented a passive biological treatment at Nickel Plate mine (British Columbia, Canada), an aerobic-anaerobic process of three different stages capable of processing 12.6 L s⁻¹ of effluents with > 1000 mg L⁻¹ of thiocyanate, reaching the final thiocyanate of 5 mg L⁻¹. The anaerobic chamber is for nitrification and denitrification, producing a higher quality effluent (Akcil & Mudder, 2003). Another industrial approach was developed by Gold Fields Limited, a pilot plant

treating wastewater with thiocyanate with an Activated Sludge Tailings Effluent Remediation (ASTER™) for process water recycling. According to Mekuto et al., 2016, its main disadvantage is the degradation of different cyanides than thiocyanate with chemical methods.

At the laboratory scale, in the past ten years (2012–2022), several studies were carried out, such as those presented in Tables 2, 3 and 4, where operational conditions (temperature, pH, agitation, cyanide source, and biodegradation-media composition) are resumed. Some bacteria degrade cyanide better in axenic cultures, while others generate cooperative relationships in consortia or co-cultures. For instance, Khamar et al. (2015) found that *Halomonas* sp. degrade cyanide better when co-cultured with a new bacterial taxon they isolated. Chaudhari & Kodam (2010) proved that *Klebsiella pnseumoniae* and *Ralstonia* sp. biodegrade thiocyanate better as a consortium, but isolated they degrade better free cyanide. In contrast, Moradkhani et al. (2018) isolated four bacteria from contaminated soil of a goldmine and compared cyanide degradation between the selected strain (*Pseudomonas parafulva* NBRC 16636 T) and the mixed culture of the four isolations, observing that axenic culture was more efficient to biodegrade cyanide than the mixed culture. Avcioglu & Bilkay, (2016) compared cyanide biodegradation with different strains of *Klebsiella pneumoniae*, observing that only one of them biodegrades cyanide better than the mixed culture.

8 Operation Modes

There are three settings for biotechnological cyanided waste treatment operation: in situ, ex-situ, or bioreactors. The first consists of biospreading, bioventilation and bioaugmentation. In situ operation modes are considered the most efficient methods of all biotreatments since they are not invasive and do not require energy; however, as they are natural attenuation processes, require indigenous cyanotrophics that resist metals and inorganic compounds, operate under environmental parameters, consider geological factors and how contaminants are distributed, which is difficult to obtain. In addition to the variability of environmental conditions (Razanamahandry et al., 2017), ex situ strategies are related to soil cultivation (solid phase treatment system), composting (anaerobic), and biopiles. They are cheap, simple methods that generate heat by themselves. They have a fast reaction rate and can be done near the area where cyanide is produced; However, they require space, and the organic material undergoes aerobic degradation, followed by irrigation and plowing (Razanamahandry et al., 2017). Bioreactor strategies previously described are classified by biomass agglomeration into suspended or immobilized biomass bioreactors. The latter can be subclassified into stationary systems or fixed beds in batch or continuous cultures. Some studies of these strategies at the laboratory level, in batches, and continuously are presented in Table 5, which summarizes their operating conditions.

Table 5 Biotechnological treatment strategies

Cyanotrophic organism	Reactor type	[CN ⁻] ₀ (mg L ⁻¹)	Operational conditions					Biodegradation efficiency (%)	References
			Agitaci ⁿ (rpm)	pH	T (°C)	O ₂ (mg/L)	Inoculum (%v/v)		
WWTP consortium	Gavel supported	Cyanide 5.8, Thiocyanate 37	NR	7–8.5	NR	NR	10.5 gL ⁻¹	99 93	Di-Biase et al. (2020)
<i>Bacillus</i> sp.	RBC	KCN, 300	5	10.5	20	NR	15	97	Guadalima and Monteros (2018)
<i>Pseudomonas parafulva</i> NBRC 16,636(T)	EF	KCN, 200	150	9.5	32	NR	1 × 10 ⁷ *	94	Moradkhani et al. (2018)
<i>Bacterial consortium</i>	Anox Kaldnes supported	Thiocyanate, 100–500 Cyanate, 60 Cyanide < 10	NR	7.4 –7.6	NR	NR	NR	98 90 95	Vincent et al. (2016)
<i>Klebsiella pneumoniae</i>	EF	KCN, 0.5** KCN, 1000	150	7	25	NR	10	88 56	Avcioglu and Bilkay (2016)
<i>Thiobacillus</i> sp.	4 aerated packed anaerobic/3 aerobic packed	NaSCN, KOCN, 165, 159	NR	7.5	25	1.5–6	NR	38.5 62.6	Villemur et al. (2015)
<i>Bacillus</i> sp.	Sequential packed	F-CN ⁻ , 500	NR	9.8	33	0	80	95	Mekuto et al. (2015)

(continued)

Table 5 (continued)

Cyanotrophic organism	Reactor type	[CN ⁻] ₀ (mg L ⁻¹)	Operational conditions					Biodegradation efficiency (%)	References
			Agitacdn (rpm)	pH	T (°C)	O ₂ (mg/L)	Inoculum (%v/v)		
<i>Pseudomonas resinovorans</i>	Culture Tubes	KAg(CN) ₂ 36.3 K ₄ Fe (CN) ₆ , 10.55	230	5	30	NR	50 nL	NR NR	Bouari et al. (2013)
<i>A. awamori</i>	Membrane	KCN, 280	Q = 3 mLh ⁻¹	>9	30	NR	NR	80	Santos et al. (2013a, b)
WWTP consortium	Continuous Flow	KCN, 10	NR	7	30	NR	2700 mg L ⁻¹	75	Kim et al. (2011)
<i>Pseudomonas pseudocaligenes</i>	Biostat B plus	NaCN, 45	NR	10	30	NR	NR	60	Huertas et al. (2010)
Bacterial consortium	Aerated clarification	NaSCN, 150	NR	7.5	30	2.3	2700 mgL ⁻¹	94	Lay-Son and Drakides (2008)
<i>Fusarium oxysporum</i>	Sodium alginate supported	CN ⁻ , 40	NR	8	30	50	NR	86	Campos et al. (2006)

RBC: Rotating biological contactor **EF:** Erlenmeyer flasks **NR:** not reported **YE:** yeast extract *: UFC mL⁻¹ **: mM **MSM:** Minimal mineral medium Q = flow for continuous cultures

8.1 Other Benefits from Biotechnological Treatments

In addition to cyanide waste treatment, some cyanotrophics metabolism could be coupled with by-product formation. For instance, Manso Cobos et al., (2015) reported a strain of *P. pseudoalcaligenes* and a mutant that accumulate polyhydroxyalkanoates (PHAs) when they grow in cyanide reactors. PHAs are biodegradable thermoplastics that show great potential to replace petroleum-derived plastics since they are environmentally friendly, non-toxic, biocompatible, and can be a useful commercial source of chiral monomers (Muhammadi et al., 2015). Microorganisms also produce pterins when they biodegrade cyanide. Pterins are essential biological molecules used for anticancer properties in chemotherapy and disease diagnoses (Kompis et al., 2005). In genomic studies of *P. pseudoalcaligenes* CECT53344, Igeño et al. (2019) reported genes with the capacity to degrade furfural and furfuran, which are recalcitrant compounds that are generated during the treatment of lignocellulosic waste to produce second-class bioethanol generation. Applying this kind of cyanotrophic could lead to the exploitation of unconventional carbon sources (which could be another environmental issue due to recalcitrance) to improve cyanide biodegradation.

9 Conclusions and Perspectives

It is essential to note that each biotechnological treatment strategy for goldmine tailings has advantages and disadvantages: selection of the cyanotrophic organism, its operational conditions, and the operational configuration need to be studied for a suitable treatment of a specific tailing composition. As with any other wastewater treatment, the tailings must be chemically characterized, and their influence on the selected cyanotrophic organisms must be tested. Although biotechnological treatments are an environmentally friendly alternative for cyanide waste treatment, they have not been fully implemented at an industrial scale due to the detailed research needed. The access to these technologies for developing regions where mines are placed must also be considered for proper biotechnological design. Nevertheless, ultimately, biotechnological treatments allow biodegradation or assimilation of F-CN^- , metalocyanides, and ammonia denitrification, giving a safe and sustainable effluent. The major challenge of biotechnological treatments is their susceptibility to environmental conditions at mines (for instance, mesophilic temperature) and the sensitivity to pollutants loading in mine tailings. For this purpose, indigenous cyanotrophics represent a way out of this disadvantage since they are adapted to such environmental conditions and pollutants with the challenge of developing specific culture techniques for those organisms.

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Chapter 11

Production of Silver Jewellery in Taxco, Guerrero, Mexico, Versus the Objectives of Sustainable Development



**María Elena Tavera Cortés, Xenia Mena Espino,
and Martha Jiménez García**

1 Introduction

Sustainable development refers to the use of available resources to meet the present needs of humanity without compromising those of the future generations through technological and institutional development (Food and Agriculture Organization of the United Nations [FAO], 2015). At the UN sustainable development summit held in 2015, more than 150 countries approved the 2030 Agenda for sustainable development. It is defined as a plan to preserve the prosperity of people and the planet, leading to a sustainable and resilient world as proposed by the 2030 Agenda for sustainable development. At the center of the 2030 Agenda are seventeen sustainable development goals (SDGs) and 169 targets with 232 indicators. These SDGs seek to end poverty, fight inequality and injustice, and confront climate change. These objectives, integrated and indivisible from one another, create a balance among the facets of sustainable development. They are considered unique because they require the application and development throughout the world (developed as well as developing countries) and standardize necessary level to combat inequalities (Kostoska and Kocarev, 2019).

It is recognized that eradicating poverty is related to the strategies to generate economic growth by involving various environmental and social requirements. On this basis, an opportunity is created for the United Nations system worldwide as well as for Mexico to emphasize cooperative participation of major groups and continue

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promoting social inclusion, equity and humanitarian equality to build responsible and conscientious citizenship. However, the objectives imply various complexities when applying economic, environmental, and social policies that positively affect the proposed objectives' fulfillment.

2 Theoretical Reference

2.1 *Sustainability Science*

Sustainability science is defined as the research in the context of socio-ecological systems (SES), also known as coupled human–environment systems or socio-ecological systems (Lange et al., 2013). SES is defined as a complex multilevel system, studied at different scales and under specific conditions. A variety of disciplines are involved in the study of SES, including sustainability science, economics, geography, and environmental science among others, which provide different views of socio-ecological interactions and outcomes. Researchers have used the concept of SES to emphasize the integral character of the relationships established between natural and social systems. Theories of analysis of socioeconomic systems have evolved to such an extent in recent decades that the problems related to the sustainable policy have been identified scientifically and practically (Binder et al., 2013). Current conditions have led to a growing need to rethink incentives and regulations in the existing systems. In this way, we can better understand the needs and refine the models of economic growth in the silver jewellery sector and promote a more equitable and friendly transition with a strategy of conservation.

One of the essential definitions of economics is that “economics is the science that studies the response to satisfy the needs of the limited order through limited resources” (Vásquez Roldán et al., 2014). A paradoxical perspective related to global sustainability is linked to economic, social, and environmental policies because they reside at diverse levels and operate through strictly separate logic and periods (Hahn et al., 2015). A correlation between environmental crisis, population growth, and expansion of economy has accelerated in recent years. This has generated a network of interdependencies between developed and developing nations to obtain tangible levels of economic development for the growing needs of the society and the provision of sustainable development. Technological innovation and development are used as a key element that helps to address sustainable development problems. Modern developments and the capacity of human beings in technological management have provided areas of opportunity to improve the sustainability of services and products (Hall et al., 2018). However, limitations persist to achieve the commitments established in the 2030 Agenda.

In the case of production of silver jewellery in Taxco in the state of Guerrero, conflicting variables converge, such as the generation of economic income derived from the commercialization of these products versus sustainable development. According to the National Institute of Statistics and Geography (INEGI), the mining–metallurgical sector in Mexico ranks first globally in the production of silver. However, this activity is having an impact on climate change, because mining is located within the ten strategic sectors having the highest greenhouse gas (GHG) emissions worldwide (Rüttinger and Sharma, 2016).

Holden et al. (2017) discussed and developed a model of sustainable development that established limitations on social and economic activities, aiming to achieve economic, environmental, and social balance. After reviewing the implementation of SDGs in several countries Allen et al. (2018), noted initial progress resulting from the pursuit of these objectives. However, they also emphasized the need for further evaluation of the interrelationships, commitments, and synergies between development and the SDGs that have yet to be addressed.

Silver is extracted from underground mines, which means that the extraction must be conducted using explosives. The explosives used for the extraction of this mineral cause the following environmental effects:

2.1.1 Changes in the Conformation of the Soil

To conduct the extraction of minerals from the surface of the soil, large excavations must be performed. Starting with deforestation in large extensions of land, excavations result in the loss of flora and fauna, erosion problems, and alterations in the biogeochemical cycles of nature, such as water, carbon, nitrogen etc (Shiquan et al., 2022).

2.1.2 Contamination of Surface Waters

In many cases, the chemical materials used or released during the mineral extraction process are not treated; so when they are discarded, they filter into surface waters, causing the death of flora and fauna inhabited in those waters. The contamination of surface waters could be also through bioaccumulation, transferred between organisms through food chains (Marshall et al., 2020).

2.1.3 Groundwater Contamination

During rains in the mining areas, wastes are carried with rainwater, and transported and permeated to groundwater deposits, causing its contamination. Water plays a preponderant role in mining–metallurgical activity; it is used for mineral processing, chemical reactions, cooling of industrial equipment, emission control, suppression of dust, transportation of tailings, and general services, such as consumption and

personal hygiene. To obtain 1 oz of silver, 115 L of water is required. The main contaminants are Zn, As, Pb, Ni and Cd, in addition to changes in the pH of water sources near silver mines (O'Neill et al., 2015).

2.1.4 Air Pollution

The extraction of minerals releases dust and other toxic gases into the environment, that are caused by the explosions used to break rocks and obtain materials. The gases released cause serious respiratory problems to humans and animals inhabiting the nearby areas. These gases move into the atmosphere, thus influencing the greenhouse effect and contributing to climatic changes. In the past, mercury was used for the extraction of silver and emitted in mining areas and their surroundings in addition to the elements that are presently released in the atmosphere (Hagan et al., 2011; Zhao et al., 2022).

To conduct a more specific evaluation of contamination generated during the extraction of silver, the process of cyanidation is used. In this process, the ground mineral is put in contact with a dilute alkaline cyanide solution, which selectively dissolves the silver, leaving the other mineral species in solid form. The mineral is recovered from the solution by adsorption with activated carbon, or by precipitation with zinc. Despite the environmental difficulty related to the use of cyanide, it is conducted because it is an economical, efficient, and, in general, easy-to-operate process (Verbrugge et al., 2021; Sonter et al., 2018).

The extraction of silver in the mines occurs as follows: The area to be explored is dynamited. The pieces go to crushers to crush stone pieces released from the explosion into smaller pieces, and passed through household sieves. Then these sieved pieces are passed to a second shredder. Once the silver is reduced to the desired size, it is shifted to mills through belts. Finally, the sampling is conducted to determine purity of the silver (Studnicki-Gizbert and Schecter, 2010).

During grinding, the silver enters steel mills, where it is converted into sludge, because it is constantly mixed with water for grinding (contamination of enormous amount of water). Subsequently, cyanide is added to the silver sludge. This is done in one continuous motion. The process of cyanidation produces foam, which is collected on flotation. This process consists of container cells and impellers that rotate, causing the silver particles to separate from the soil and ground stone. In this way, the silver manages to float in the form of foam. Thus, the foam is released through lateral ducts and sent to foundry. Here, the collected foam is placed in canvas bags, where it is compacted through a pressure process. Because of this exerted pressure, it is possible to divide cyanide water and anode sludge, allowing the sludge to deposit for smelting.

The anode sludge is melted in furnaces. This smelting is conducted using diesel or petrolatum (which is an occlusive moisturizer with a lower degree of purity). Once the silver anode mud plates are obtained, they are deposited in vats; the plates are disintegrated by chemical and electrical processes. In this way, silver crystals, called

silver shots, are derived. Subsequently, silver grits are shifted to furnaces to melt the said material.

In the final process, once the silver is melted, it is deposited on rotating plates that rotate horizontally around the furnace.

Mining operations require a significant amount of electrical energy and fuels for extraction, grinding, and processing of minerals. These operations use fossil fuels, such as natural gas, fuel oil, metallurgical coke, petroleum coke, gasoline, diesel, LP gas, as well as electricity from conventional and alternative sources (Camimex, 2020).

2.2 *Methodology and Analysis*

A diagnosis is made with economic variables to identify correlations between the objectives of sustainable development and the production of silver jewellery in Taxco, Guerrero.

2.2.1 *Mining in Mexico*

According to the data released for 2019–2021 by the United States Geological Survey (USGS, 2022), mining production for Mexican economy has economic, social, and cultural–political relevance. Globally, Mexico ranks number one in the production of silver with an average share of 27.5%; China is second with 16.46% share, and Peru is at third position with 16.03% share. For 2019–2021, the global concentrated production of silver for these three countries was 59.9% (Fig. 1). According to the INEGI, in 2021, the contribution of mining–metallurgical sector in the Mexican economy was 8.3% of the industrial gross domestic product (GDP) and 2.3% of the national GDP, with an investment of \$3,532.62 million. According to a report of the Mexican Social Security Institute (IMSS), the mining–metallurgical sector generated 367,935 direct jobs in 2021.

The environmental impact, according to the Ministry of the Environment and Natural Resources (SEMARNAT), is defined as “the modification of the environment caused by the action of man or nature.” According to another definition, environmental impact is defined as the “impact that implies adverse effects on ecosystems, climate, and society due to activities, such as the excessive extraction of natural resources, the inadequate disposal of waste, the emission of pollutants, and the change in land use, among others.”

As seen in Fig. 2, during 2013–2021, the proportion of silver recycling was at an average of 15.56%, which implies a positive action for the environment.

Because raw materials are obtained more efficiently by reducing carbon footprints, silver recycling represents an opportunity to reduce degradation under the United Nation’s 2030 Agenda. In the Agenda, goal 8 is for economic growth and goal 13 is for climate change, both objectives converge on the importance of recycling to have

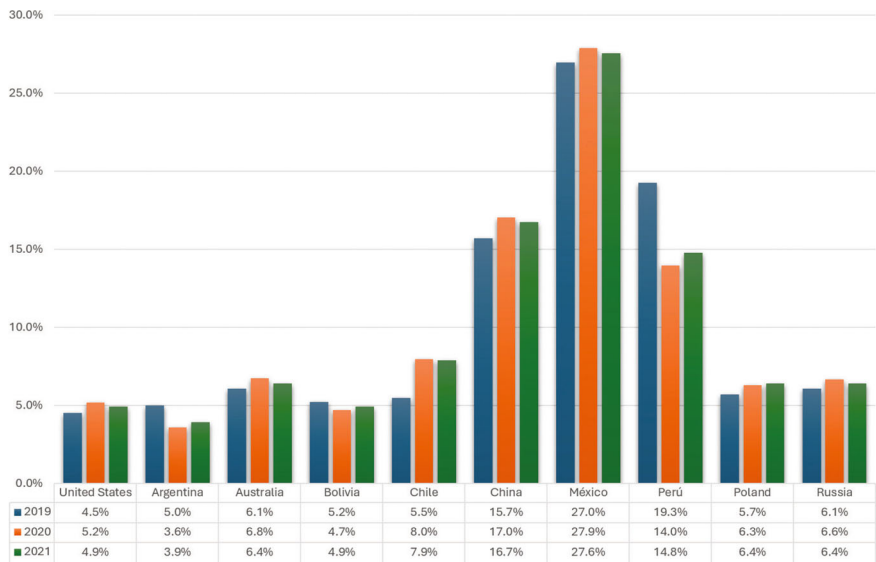


Fig. 1 The percentage share of global silver production. (Source: USGS, 2022)

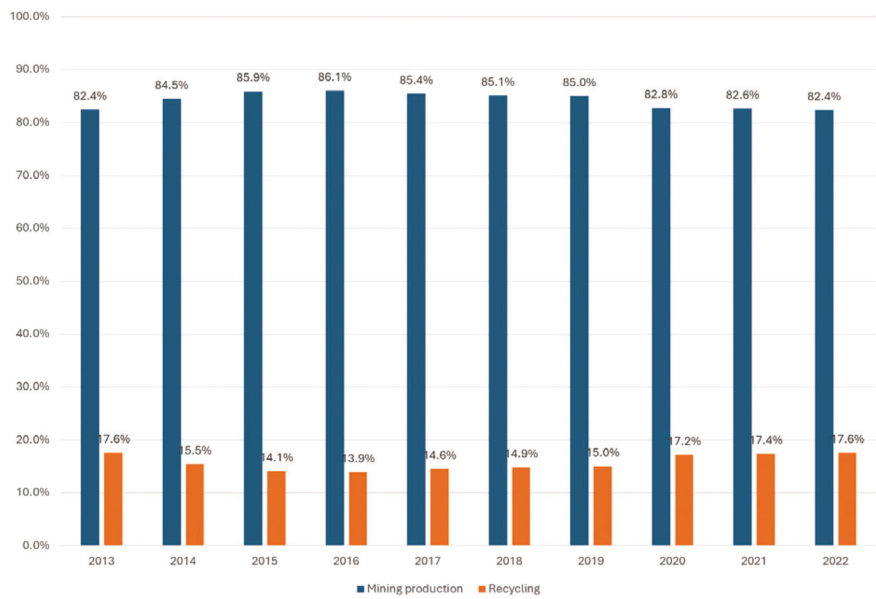


Fig. 2 Percentage of silver production versus recycling. (Source: USGS, 2022)

a positive impact on the environment. The environmental impact is related to the protection of the environment, which is a requirement for sustainable development, considered to promote economic growth, social equity, and environmental protection.

Sustainable development requires the environment to be stable and healthy, there is no negative environmental impact, and the population grows and obtains resources in a stable and healthy manner. An issue related to the environmental impact is that of environmental costs and, more specifically, mining–metallurgical activity, which has a greater impact on the contamination of abiotic media, usually related to water, soil, subsoil, air, and sediments. Currently, there is a global environmental impact related to the economic activity of mining (Trivedi and Chakraborty, 2001). There is also the issue of restoration of contaminated sites, which pursues reducing the dispersion of pollutants through the mining–metallurgical industry. Worldwide, the mining industry is identified as one of the main sources of the generation of heavy metals polluting the environment (Yang et al., 2012). The foregoing result in the affectation of both terrestrial and aquatic ecosystems, in addition to changes in the landscape and destruction of agricultural areas, soils, and the so-called acid mine drainage, which is the result of mining waste deposits. In addition, there is the affectation of abandoned mines and air pollution caused by suspended particles, emitted by the transport used for the economic activity of mining. Within the impacts of mining, are the effects on the atmosphere in the natural environment (Stansfeld and Matheson, 2003).

The state of Guerrero, previously being a symbol of mining, today does not contribute a significant amount to the national production of silver; however, it has seven mining districts, among which Taxco stands out, as shown in Fig. 3. Guerrero is also one of the districts that has a wide variety of minerals to be exploited. This state, with the city of Taxco de Alarcón, was at position four among the silver-producing states of the national territory. However, more recent studies, including the one published in 2020 by INEGI and the general directorate of mines, indicate that their share of silver production has fallen. Currently, it is estimated that more than half of the silver produced in Taxco is transformed into jewellery and other products derived from it; most of the produced silver is used in informal production, even when it is marketed formally. Likewise, it is estimated that more than 70% of Mexican silver jewellery exports come from Taxco (Mexicanarte, 2021). At least 15,000 artisans who are officially dedicated to goldsmithing, carving, sculpting, and embossing silver pieces are employed in this town (Guadarrama-Serrano, 2009).

Silver jewellery is extremely important for the local economy, because, except for agriculture, it concentrates the largest number of employed personnel, excluding those who are only dedicated to the sale of jewellery and are included in “retail trade” and “informal trade.” According to some estimates, up to 30,000 personnel are employed in this sector, as the official records do not include clandestine workshops or the work done by minors (Yeganeh, 2020).

According to estimates by the National Chamber of the Silver and jewellery Industry, around 400 tonnes of silver is consumed annually for transformation at the national level, of which around two-thirds are for jewellery and goldsmithing. Globally, the main use of silver is for industrial purpose and to a lesser extent for



Fig. 3 Map of the location of Taxco in the state of Guerrero, Mexico. (Source: Secretary of Mining)

jewellery, although short-term expectations indicate that in the future, the greatest demand will be for jewellery. Regionally, Taxco is the most important area for silver consumption for the purpose of jewellery. The productive chain of silver jewellery is shown in Fig. 4; it begins with the acquisition of raw material direct from producers or resellers. The production is conducted in large workshops or through the maquila scheme, in which marketers and artisans participate. Marketing is conducted from producers to wholesalers, and from there to retailers or directly to the final market. A small part is marketed directly from producers to consumers.

The main problem with mining is the dumping of heavy metals in the environment because the total mass of dumped metals is preserved. This entails a serious problem of environmental pollution because the concentration of dumped materials in mining areas is higher than the permissible limits, thereby causing a risk for both flora and fauna and the population living in that geographical and surrounding areas. As

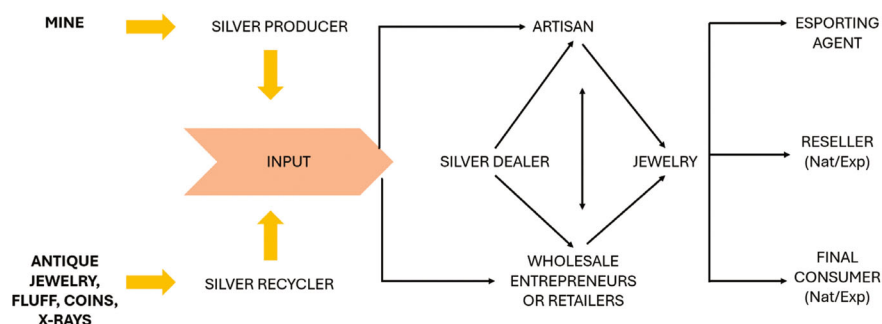


Fig. 4 Chain of production and commercialization of handcrafted silver products in Taxco, Guerrero. (Source: Secretary of Economy of the State of Guerrero, Report 2018)

Taxco de Alarcón is a historically important site and identified as a mining center, its ideological current, which is in the region called Rio Taxco, receives the largest amount of wastewater in the entire municipality as well as from other settlements that surround the municipality.

Studies are conducted on the environmental impact of economic activity of mining, focusing on the river Taxco in the Taxco area of Guerrero. The results show that tributaries of the river Taxco are contaminated by heavy metals because of tailings and high concentration of lead (Salcedo et al., 2022; Zheng et al., 2020) along with higher presence of elements, such as iron, zinc, cadmium and copper (Mendez-Ramirez and Aurora Armienta, 2012).

Metals such as iron, zinc, copper and cadmium cause higher risk to the environment because the species affected by their contamination tend to have a greater affectation by metal toxicity. Therefore, it is recommended that in order to reduce environmental impact, acidity of the water be regulated in addition to reducing the concentrations of these metals in the environment.

Another problem is that tailings and acid drainage impact the environment. This can be reduced by using physical barriers so that metal particles affecting the environment do not reach passive treatment systems used for acid drainage by the municipality of Taxco de Alarcon (Salcedo et al., 2022).

It is worth mentioning that the contribution of the tributary of the Taxco River does not correspond to the contribution of metals that this has to the tributaries to which they feed because its flow is lower with the contributions of metal that it has. With these studies, it is concluded that it has high concentrations, which implies that types of the river contain water that is not suitable for human consumption, mainly in the nearby sites on which that river depends, which implies that it is essential that this type of water be treated before being discharged or deposited in the Taxco river which will later feed other tributaries (Verbrugge et al., 2021).

The objective of the 2030 Agenda is to reduce negative impacts on the environment. Based on this premise, it is important to establish the indicators that allow the production of silver jewellery in Taxco, Guerrero, and reconcile the same with positive signs for a less carbon-intensive transition that align with SDGs.

3 Conclusion

This study elicits the importance of proper management of silver production in order to establish positive signals for a less carbon-intensive transition. Although the use of silver in the jewellery sector of Taxco represents its importance for generating employment, which positively affects the income of families involved to the commercialization of silver, it also implies deterioration of the environment. This deterioration could be minimized by using silver recycling for the jewellery activity and having income from its sale by aligning with the objectives of sustainable development.

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Chapter 12

Decarbonisation Policies in Mexico



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Abstract Decarbonisation is known as low-GHG emissions development, and while it is unclear precisely what that entails in these circumstances, however, we can refer to the definition stated by the Intergovernmental Panel on Climate Change (IPCC) mentioned in its fifth report: Process by which countries or other entities try to achieve a down economy carbon emission, or by which people try to reduce their carbon consumption (IPCC. (2014). Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, In Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.) IPCC, Geneva, Switzerland, pp. 151.).

Abbreviations

CEC	Commission for Environmental Cooperation of North America
CCS	Carbon Capture and Storage
CFE	Federal Electricity Commission
CONAFOR	National Forestry Commission
CRE's	Energy Regulatory Commission's
DDPP	Deep Decarbonization Pathways Project
LIE	Law of the Electricity Industry in Mexico
GHG	Greenhouse Gas
INECC	National Institute of Ecology and Climate Change
IPCC	Intergovernmental Panel on Climate Change

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SAGARPA	Secretariat of Agriculture, Livestock, Rural Development, Fisheries, and Food
SEMARNAT	Secretariat of Environment and Natural Resources
SDSN	Sustainable Development Solutions Network
SLCP	Short-Lived Climate Pollutants
UNFCCC	United Nations Framework Convention on Climate Change

1 Introduction

Decarbonisation is known as low-GHG emissions development, and while it is unclear precisely what that entails in these circumstances, however, we can refer to the definition stated by the Intergovernmental Panel on Climate Change (IPCC) mentioned in its fifth report: Process by which countries or other entities try to achieve a down economy carbon emission, or by which people try to reduce their carbon consumption (IPCC, 2014). According to (Rueda Abad, 2019), the definition of decarbonization is very general; nevertheless considers that there are two points critical to highlight, the first is that we are talking about state economic processes and it is not precisely the States that generate the emissions but also the companies, which as a whole become the main emitters of Greenhouse Gas (GHG), so programs and public policies should be proposed whose tendency is to reduce emissions. The second aspect involves individuals and other subnational actors being aware of their coal, which should also contribute to reducing greenhouse gases. These two important considerations set the tone for the IPCC to give a definition clear for decarbonization in its Special Report of 2018: *The process by which countries, individuals or other entities pretend achieve an existence zero fossil carbon usually refers to a reduction of carbon emissions associated with electricity, industry and transport* (IPCC, 2018).

Over the past ten years, there has been an increase in consideration of the effects of climate change reciprocated with the mining sector focused on adaptation towards climate change problems. In accordance with the 2015 Paris Agreement, 195 nations committed to keeping global warming well below 2.0°C and, ideally, at or below 1.5°C over preindustrial levels. If that goal were to be achieved, it would lead to a decarbonization of all industries, which would cause significant changes in the demand for mining-related commodities and probably result in diminishing world-wide mining revenue pools. The appraisal of mining portfolios must now consider the prospective decarbonization of other industries.

Currently, 4–7% of the world's greenhouse gas (GHG) emissions are from mining. Scope 1 and Scope 2 CO₂ emissions from the industry (those brought on by mining activities and electricity use, respectively) total 1%, while fugitive methane emissions from coal mining are estimated to be between 3% and 6%.

This chapter will analyse the decarbonisation policies in Mexico's mining industry, looking at the methods and actions taken to encourage ethical mining practices and help reduce greenhouse gas emissions.

2 Regulatory Framework of Mining in Mexico

Natural resources are established as national assets under Article 27 of the Political Constitution of the United Mexican States, implying that the Federal Government is responsible for establishing the terms for their utilization. Our nation’s mining regulations are derived from this article, although several secondary laws were added to this activity’s regulatory framework. The following laws and rules pertain to mining in Mexico now (Fig.1):

The Ministry of Economy, through the Unit of Coordination of Extractive Activities (previously the Under Secretariat of Economy), encourages sustainable and socially responsible mining, respecting all life forms, including human, animal, and plant life, through the generation and dissemination of Best Practices Guides and Compliance Protocols as follows:

- Guidelines for Best Mining and Metallurgical Practices: Sanitary Contingency 2020 SARS-CoV-2 (COVID-19); includes the declaration of a health emergency, suspension of general activities, definition of operational disciplines for the conduct of essential activities, issuance of general guidelines by labour and health authorities, and announcement of the strategy for resuming activities on May 13, 2020.

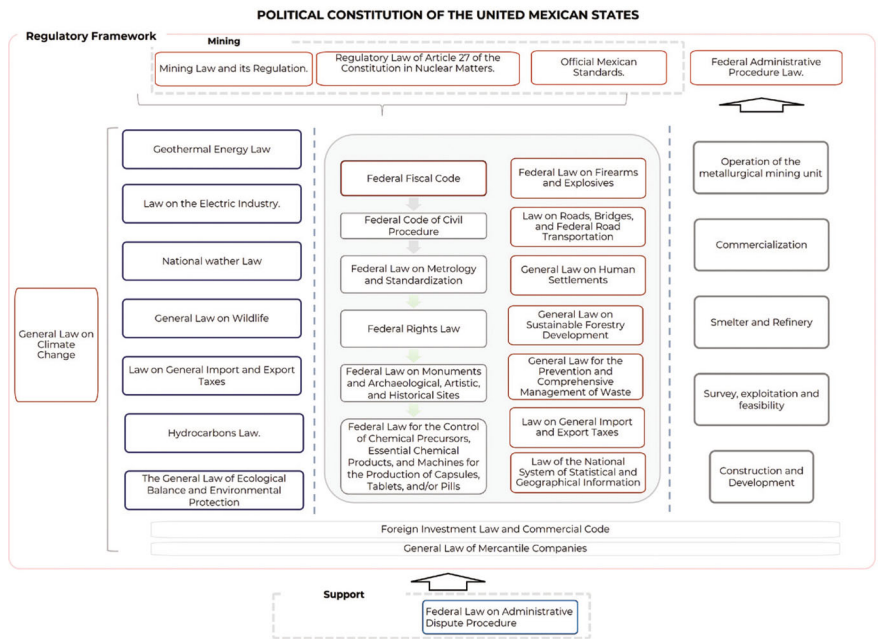


Fig. 1 Laws and regulations for mining activity in Mexico.

- **Community Relations Guide:** enables mining operations to comply with the Constitution and laws by promoting inclusiveness, respect for human rights, gender equity, protecting children's best interests, non-discrimination, environmental protection, and the common good.
- **Water Use and Management Guide:** A mining operation's primary studies and licenses must adhere to water.
- **Basic Guide: The Protocol of Good Finance Practices in the Mining Sector** lays forth certain broad guidelines, whose observance must, therefore, help in enhancing relations between mining enterprises, the tax authority, and the federal mining authority.
- **Guide for Tailings Management:** demonstrates the laws and rules that must be implemented to ensure tailings' stability and environmental performance at disposal sites.
- **Guide to Good Practices in Mining and Community Relations:** outlines the ideal methods for mining firms to interact socially with locals at various stages of their producing activities.
- **Protocol for Public Consultation in Mining:** focuses efforts on a more equitable activity, including communities and their environmental effects.
- **Protocol for Security and Asset Protection in the Mining Sector:** enhances the mining industry by advocating the adoption of appropriate security procedures through a culture of risk prevention.

3 Law of the Electricity Industry in Mexico (LIE)

This law, enacted on August 11, 2014 and published in the Federation's Official Journal, included clean energy and/or renewable energy production as one of the co-generation of energy items to contribute to the energy mix at the time.

- Wind
- All kinds of solar radiation
- The various types of marine energy, including tidal, thermal, waves, ocean currents, and the gradient in salt concentration.
- The geothermal deposits' heat.
- Products classified as bioenergy under the Law for the Promotion and Development of Bioenergy Products
- The energy produced by using the calorific power of methane and other related gases at waste disposal facilities, animal farms, and residuals in water treatment plants, among other places.
- The Energy Regulatory Commission's (CRE's) minimum efficiency requirement and the Ministry of Environment and Natural Resources' emission standards must be managed to reach for the power generated for the use of hydrogen for use in fuel cells or combustion, respectively.
- Hydroelectric power plant's electricity
- Nuclear power energy

- Energy produced using waste processing byproducts such as agricultural skimming or urban solid waste (such as molecular gasification or plasma), as long as the processing does not produce dioxins, furans, or other emissions that can seriously affect human health or the environment, thus it needs to comply with official standards set by the Ministry of Environment and Natural Resources of Mexico (*in Spanish known as SEMARNAT*)
- The energy produced by cogeneration facilities that meet the CRE's efficiency standards and SEMARNAT's emission standards.
- The energy produced by sugar refineries corresponds to SEMARNAT's emissions standards and CRE's efficiency standards.
- Energy produced by thermal power plants that encapsulate and store carbon dioxide through geological processes or biosequestration and have an efficiency of at least equal to or greater than that in terms of kWh produced per ton of carbon dioxide equivalent released into the atmosphere and meet the emission standards set by SEMARNAT.
- Technologies recognized as having minimal carbon emissions by international criteria standards.
- Other technologies are approved by SEMARNAT based on criteria and requirements for energy and water efficiency, emissions into the atmosphere, and waste generation, directly, indirectly, or throughout a product's life cycle.

On February 3, 2021, the Head of the Executive Power presented to Congress a preferential initiative to reform the existing LIE, prioritizing the Federal Electricity Commission (CFE) over private industry, (PWC, 2022). The reform was approved by the Legislative Power, so on March 9, 2021, it was published in the *Diario Oficial* of the Federation. However, it did not enter into force due to a final suspension granted to the Power Court of the Federation on March 19, 2021. To date, power generation clean or renewable has remained practically reduced to the minimum expression in our country, even when the deep decarbonization model of the Mexican energy system by 2050, presented in July 2015 by the INECC, to contribute to the COP21 in Paris, contemplated the following objectives:

- Explore the technical feasibility of reducing Mexico's greenhouse gas emissions by 50% by the year 2050, compared to the levels of the year 2000, in accordance with the objective established by the General Law on Climate Change, and so consistent with the goal of not exceeding 2°C.
- Investigate the nature, scale and timing of the changes required to achieve this transformation, as well as the main challenges and opportunities.
- Provide the means and perspectives to evaluate the evolution of the necessary policies to achieve the expected objectives in relation to deep decarbonization by the year 2050, in a credible and effective manner, taking into account the various costs that this implies.
- Identify opportunities for detailed deep decarbonization analysis with the aim of suggesting future policy and investment decisions.

4 Paris Agreement and Commitments of Mexico

According to Christiana Figueres, Secretary of the UNFCCC Executive, the Paris Agreement affirms the world's unstoppable shift toward a low-carbon, secure, and healthier future.

The Conference of the Parties 21, held in Paris from November 30 to December 12, 2015, brought together 195 countries and nearly 45,000 participants, making this event the most important meeting on climate organized to date. However, the Nationally Determined Intended Contributions (NDCs) presented were not ambitious enough, since the aggregate levels of annual global emissions would not allow the 2 °C goal to be met by the end of the 21st century. Therefore, it was necessary for the Agreement to be reached to include adaptation actions and concrete implementation measures that allow highly vulnerable countries to face the adverse effects of climate change. The event led to the adoption of the Paris Agreement, signed by 175 countries on April 22, 2016, in New York.

This agreement recognizes, among other things, the importance of adaptation as well as the vulnerability of certain developing countries. It describes the commitments established in Article 7, and to finance its actions, it provides for the contribution to the Green Climate Fund of one hundred billion dollars per year by developed countries, while developing countries may contribute voluntarily. This is the case of Mexico, which announced its contribution to said Fund on September 5, 2015, and, by signing the Agreement, reaffirmed its commitment to act on climate change. Since the creation of the United Nations Framework Convention on Climate Change (UNFCCC), Mexico has incorporated into its legal and institutional framework the respective commitments that it implies. It began the design and implementation of related policies. However, concrete adaptation actions in the country, even in the most vulnerable areas, are frequently relegated to other priorities (Iniciativa Climática de México, 2021; Muñoz et al., 2016).

Five of the Paris Agreement's ten components are considered essential to supporting low-emission development and adapting to climate change.

- The primary goal is to keep the planet's temperature increase below 2°C by 2020 when this century will be at its conclusion.
- Each country evaluates and bolsters its national contributions every five years (NDC).
- Climate for supporting developing nations at \$100 billion (from 2025).
- Mechanisms for accountability to guarantee that the requirements are achieved (until 2020).
- The main strategy to support vulnerable nations is adaptation.

5 Commitments of Mexico

Unconditional reduction recommends reducing greenhouse gas emissions (GHG) and Short-Lived Climate Pollutants (SLCP) by 25%, or 22% of E3 GHG and 51% of carbon.

By producing 35% clean energy in 2024 and 43% in 2030, the industrial sector could perhaps reduce its emissions. The scenario that its execution is not possible at this time and with the national strategy being developed. Our nation asserted: “Mexico will manage to reduce the carbon intensity of GDP by 40% and will have a maximum emissions peak around the year 2026.”

According to statistics provided by the INECC, Mexico’s emissions of 665,304,92 Gg of CO₂ in 2013 represented 1.4% of world GHG emissions, placing us at position 12 amongst some of the issuing countries on a global scale. Transportation, power production, industry, oil and gas, agriculture, land use, waste (solid urban, water residual), and residential sectors all contribute to GHGs on a national scale, with corresponding rates (in %) of 26.2, 19.7, 17.3, 12.1, 12, 4.9, 4.6, and 3.9 respectively. (IMCO, 2016; Stocker, 2015).

In terms of adaptation, the commitments are:

- Among other factors, the social sector needs to adapt to climate change, which assures food security and access to water, restricts the number of municipalities at peril by half, and involves civil society in creating public policy.
- Ecosystem-based adaptation aims to maintain and restore ecosystems while reducing deforestation to zero% by 2030.
- Strategic infrastructure adaptation, which guarantees and aids in monitoring the treatment of urban and industrial wastewater in populated areas with more than 500,000 people (IMCO, 2016).

To concentrate on the issue that concerns us, we will outline the key difficulties and impediments to executing these initiatives for each area. And considering the signing and ratification of the Paris Agreement, are these the routes found to get around the challenges?

5.1 Energy

What is the nation’s plan for its energy structure in 2050? How does the nation plan to deploy carbon capture, storage, and utilisation technologies? Are negative emissions going to be introduced? For instance, using biomass energy with Carbon Capture and Storage (CCS) or soil carbon. Mexico recently reformed its political constitution with respect to the energy sector. This allowed the participation of foreign companies in the market. The government believes that it can boost this sector, which should lead to an increase in emissions. At the same time, however, it has pledged to reduce

emissions from the sector, through developing renewable energy and promoting “clean” practices.

Emission reduction offsets in this sector have been achieved through:

- Contributions made in 2015 that were nationally determined contributions (NDCs) to the United Nations Framework Convention on Climate Change
- The Declaration of the North American Leaders on the Alliance for Climate, Energy and the Environment
- The National Policy for reducing methane emissions in the oil and gas sector (under development).

In addition, Mexico has adopted an Energy Transition Plan, which includes a comprehensive decarbonisation vision for the 2015-2020 period, a conducive and coherent institutional framework, and coherent national sectoral planning for the 2020-2025 period, and implementation and monitoring—coordinated for the period 2025-2050 (Fig. 2; LIE, 2014).

The three primary pillars of Mexico’s deep decarbonisation plan are low-carbon power, electrification, and changing the fuels used for final energy usage in all sectors.

The first of these pillars is the first milestone for deep decarbonisation. SEMARNAT is investigating CO₂ capture and storage systems, and the report “Geological Storage of CO₂” was issued in 2014 by the Secretary of Energy. Similarly, SEMARNAT is in talks to establish a national greenhouse gas market or consider Mexico’s participation in international markets.

2015-2020. Comprehensive vision of decarbonization:	2020-2025 Leading and consistent Institutional Framework:	2025-2050 Coordinated instrumentation and monitoring:
• Energy efficiency and security • Mobility and urbanism • Freight corridors and distribution alternatives • Zero GHG electricity and energy storage/smart grids • R&D in key technologies (CCS, vehicles, photovoltaics, etc.)	• Regulatory and policy market instruments • Sectoral planning consistent with the national vision • Decision between alternative routes • Influence supply and demand Investments and infrastructure • Evaluation metric	• Strategic Subgroups Example • Industry+freight transport+commercial • Urban structure+passenger transport • Electricity eneration+ distributed system integration+demand management+electric vehicles

Fig. 2 Main pillars of decarbonisation in Mexico. (INECC, 2015; LIE, 2014).

5.2 *Building*

Despite the agreement reached within the Commission for Environmental Cooperation of North America (CEC) regarding the renovation of government buildings and the New Construction Law 2008, actions in which Mexico has not yet participated, there is no real policy regarding the construction of buildings in Mexico.

The requirements for sustainable construction have improved during the past five years. For instance, as per the statement mentioned by SEMARNAT in the Mexican Norm -NMX-AA-164-SCFI-2013 “Sustainable Building, Criteria and Minimum Environmental Requirements” and the one being prepared by the National Institute of Educational Infrastructure for “sustainable schools” (INIFED). Energy efficiency is a key component of both standards, which will aid in lowering greenhouse gas emissions in the industry.

However, during the past 15 years, Mexico City has significantly increased the number of surfaces with sustainability certification. In 2010, there were 31,912 square meters of LEED-certified buildings in Mexico City, which is approximately 40% of the rental space in Reforma’s Tower. Mexico City already had 1,059,000 square meters in 2014. (33 times more than in 2010). This space will be increased in 2016 to 1,600,000 m², 52 times larger than the tallest tower. It is significant to note that many of these buildings are designed to house the headquarters of global organisations. Due to various causes, including changes in legislation and an inherent understanding of the impact of climate change and its repercussions, the number of sustainable buildings has not significantly risen.

5.3 *Transportation*

What are the sectoral objectives that the country has set for 2050 (and possibly intermediate to 2030)? What are the structuring levers that will allow them to be achieved? (energy efficiency of vehicles, load rate, modal shift, carbon intensity of fuels, variation in mobility demand) What vision does the country have of the structure and level of performance of its light vehicle fleet by 2050? Moreover, what developments and measures will allow this level of performance to be achieved (emission targets for new vehicle sales, technological choices, etc.)? What dynamics are foreseen from the point of view of mobility demand?

The transportation system in Mexico is the main source of CO₂ emissions, which includes private transportation, aviation, maritime transportation, and rail transportation. According to the V National Communication of Mexico presented to the United Nations Framework Convention on Climate Change (INECC, 2012), emissions from the transport sector contributed 46% of the 70.4Gg of national emissions of black carbon in 1990, and in 2010, the contribution increased to 54% of the total of 104.5 Gg according to a SEMARNAT report in 2012.

Given the increasing tendency of the personal automobile, the sector's emissions will increase from 67 million tonnes of CO₂ in 2008 to more than 347 million tonnes of CO₂ in 2030, with 72% of the emissions corresponding to heavy and light private transportation—choices for mitigation. The three basic pillars of decarbonisation can be used.

- Firstly, construct cities that are more compact and well-connected by carefully planning and building them. Reduce energy use by developing an urban transportation infrastructure and reducing journeys in private automobiles to satisfy the rising demand for transportation. To meet deep decarbonisation goals, an intense program of transportation infrastructure expansion and new urban design must be developed by 2025. This transition implies a significant investment and a shift in residents' behaviour.
- Following this, the widespread use of electric cars is encouraged by the use of numerous technological advancements.
- Finally, the reduction of carbon emissions in the power-generating sector will be impacted by technology choices made in the transportation sector. Enhanced reliance on renewable energies in the energy matrix requires intelligent "smart-grid" network architecture.

5.4 *Agriculture*

What variation in total agricultural production is expected, associated with the emission reduction commitments assumed in the strategy? What structural changes are expected? (in particular, distribution of livestock/extensive crops, dynamics of livestock intensification/extensification) Mexico, a country with a long agricultural tradition, has an area of 198 million hectares, of which 145 million are dedicated to agricultural activities, the main activities in the country's rural areas, where nearly 22% of the national population lives. The total emissions of this sector in Gg of CO₂ during the period 1990-2010 were an average of 89,161 tonnes of CO₂ in 1990 and 92,184 tonnes of CO₂ in 2010, according to figures announced by the INECC in 2013.

The Ministry of Agriculture, Livestock, Rural Development, Fisheries, and Food (SAGARPA) has identified six mitigation measures that need to be established in the agricultural sector at the national level:

- Water and energy conservation for irrigation technology.
- Modernization of conventional agriculture that is sustainable.
- Arable land preservation and sustainable land use techniques.
- Reconversion is productive.
- When the sugar cane is still green, it should be harvested.
- Usage of fertilisers as necessary.

The main problem with these programs is the lack of follow-up, which causes great uncertainty in the sector.

5.5 *Forest*

What evolution of the forest sink is expected up to 2050 or beyond? What service handles these questions? To what extent will the use of biomass allow the objectives to be achieved? More generally, what are the perspectives on the dynamics of non-food uses of biomass?

According to figures from the National Forestry Commission (CONAFOR), 73% (142 million) of the 190 million hectares of the national territory have a diverse vegetation cover, pine forests, oaks, ombrophilous forests, humid and dry forests, and arid zone.

For Mexico, reducing greenhouse gas emissions from deforestation and forest degradation and conserving and increasing forest carbon reserves is a necessary and priority strategy to achieve national climate change mitigation goals. Unfortunately, climate policy in this sector still has limits today, reflected in its secondary position on the national agenda.

Two basic carbon mitigation options are recognized in the sector:

- Conservation of forest biomass, protection of natural areas under protection, encouragement of sustainable management of natural forests, promotion of renewable wood consumption, and reduction of fires.
- The growth of commercial wood, paper, and rubber plants, among other things, along with energy plantations and agroforestry systems, as well as planned reforestation to rehabilitate damaged regions.

Last but not least, Mexico aims to enhance its Law for Sustainable Forest Development through the REDD+ alliance (Reducing Emissions from Deforestation and Forest Degradation).

5.6 *Industry*

What emissions trajectory is anticipated for the industry by 2050? What regulatory measures (emissions standards, taxes, carbon market) are applied to the different industrial sectors?

The industrial sector contributes 30% of global greenhouse gas emissions, according to the 2014 IPCC report. In Mexico, the sector's emissions come from mineral products, the chemical industry, metal production, and the production and consumption of halocarbons and sulfur hexafluorides (INECC, 2012).

Mexico has made specific proposals for the mitigation of greenhouse gases of industrial origin, such as high-efficiency burners, the improvement of existing systems, and even the change of fuels. One of the leading mitigation options is to increase the use of renewable energy instead of fossil fuels. Today, the sector faces a severe financial and investment crisis, limiting better returns and achieving mitigation objectives.

- Effective ways are found to reduce GIE emissions through:
- Diagnosis through the emissions inventory
- Information on business performance.
- Cleaner production
- Comprehensive environmental management systems
- Government and industrial agreements.

5.7 Waste

What is the expected evolution of the volume of waste? What changes are planned for the recycling fee?

The top initiatives put forth by this industry to lower GHG emissions include:

- Encourage the private sector to get involved in recycling initiatives, trash separation, reuse, waste containment and collection centre development.
- Create policies and guidelines that will make businesses share responsibility for managing the garbage that has been generated.
- To stimulate investment in technical and logistical advancements for improving the fee structures for collection and treatment services.
- Enhance public awareness initiatives to educate people about the value of conserving water and minimising trash output.
- Sewage treatment can profit from decreasing GHGs and using treated water for various purposes and methane to produce power.
- Between 2002 and 2011, the capacity of recycling—the collection and use of biogas produced by landfills and wastewater treatment plants—increased by 154%, creating significant potential for the industrial sector because of the cost reductions.
- Composting, which might significantly boost agricultural output.

The first element that Mexico must have is continuous, transparent and practical information that guarantees reliability through quality records and allows its uncertainty to be estimated, achieving it with laws on the matter to move towards the fight against climate change, as is the case of developed countries.

6 Carbon Price

What place can be seen in the carbon price strategy (market or tax) at the national sub- or supra-national level? What is the carbon price outlook?

During the validity of the Kyoto Protocol, Mexico has had the characteristic of not having legally binding numerical reduction commitments. This forced it to participate in clean development mechanisms, selling emission reduction certificates to developed countries. The CDM makes it possible to negotiate in the market certified CO₂

reductions generated by projects such as the production and use of renewable energy, the management and disposal of solid urban and agricultural waste, the destruction of industrial gases, reforestation, the efficient use of energy and transportation, among others. It is from the COP21 that all countries are obliged to apply measures that lead to reduced emissions, causing Mexico to seek to enter the new carbon markets.

6.1 *Carbon Neutral*

A balance between sources and sinks of greenhouse gases is set forth as a goal for the second half of the twenty-first century in Article 4.1 of the Paris Agreement. The topic of carbon neutrality is evolving quickly and is starting to be mentioned in long-term plans. This carbon neutrality can be defined from an accounting perspective by using foreign credits (funding of mitigation measures in other countries), depending on the time horizon examined, or strictly (this is the long-term objective at the global level, even if the exchanges between countries are still potentially possible). The Mexican decarbonisation policy, elevated to the year 2050, does not explicitly address this issue.

6.2 *Modeling and Impact Analysis Tools*

Is the strategy accompanied by detailed scenarios quantified by sector up to the horizon of the strategy? What scenarios were used to carry out the prospective modelling exercise on which the strategy is based? What types of models were used? Are the interactions between sectors considered (for example, the impact of agriculture's evolution on land use change)? Has an evaluation of the macroeconomic and redistributive impacts of the strategy been carried out?

The INECC has developed sectoral scenarios for the reduction of CO₂ by 2050. To develop these scenarios, common tools and methodologies were used in the 16 countries that contribute to the Deep Decarbonization Pathways Project (DDPP) and are responsible for more than 70% of the global GHG emissions.

7 Conclusion

The yearly CO₂ equivalent (CO₂) emissions from the mining sector range from 1.9 to 5.1 gigatons. Between 1.5 and 4.6 gigatons of fugitive coal-bed methane are emitted during coal mining, mostly during underground activities, and are the principal source of emissions in this industry (IPCC, 2014). The mining sector's use of electricity produces 0.4 gigatons of CO₂. Any genuine attempt to carry out the objectives of the Paris Agreement would demand a significant commitment from the whole value

chain. All sectors must cut CO₂ emissions from 2010 by at least 50% by 2050 to stay on track for a global 2°C scenario. A decrease of at least 85% is likely required to keep warming to 1.5°C. The reported emission plans for mining corporations are typically more restrained than that, with low targets, no targets specified until the early 2020s, or an emphasis on emission intensity rather than absolute numbers. Many variables, including the involvement of investors, regulators, and customers, influence external pressure to decarbonise. The level of decarbonisation will also differ by region, market sector, and executive goals. Companies should disclose their “transition risks” under a 2°C decarbonisation scenario, according to the Task Force on Climate-related Financial Disclosures (TCFD), a partnership with backing from more than 300 investors and approximately \$34 trillion in assets under management.

The coal industry would need to significantly decarbonize the mining sector, especially in addressing fugitive methane. It is possible to capture and use methane to produce electricity, although it is not a typical practice. For instance, the Coalbed Methane Outreach Program of the Environmental Protection Agency in the United States collaborates with the coal mining sector to encourage project development and remove technical and other implementation obstacles. However, not every type of mine has a ready-made answer, and often the cost is not worthwhile.

Among other things, the decarbonisation potential of mines varies depending on the product, mining type, power source, and grid emissions. Noncoal miners might fully decarbonise by utilising several levers across the sector. Some are more cost-effective than others; operational effectiveness, for instance, can gradually reduce the energy intensity of mining operations. Even in off-grid settings, switching to renewable energy sources for power is becoming more and more viable since the price of battery packs is anticipated to drop by 50% between 2017 and 2030. Electrifying mining equipment, such as diesel vehicles and gas-guzzling appliances, is becoming cost-effective. Only 0.5% of mining machinery is entirely electric now. However, in some instances, compared to conventional internal combustion engine cars, battery electric vehicles’ total cost of ownership is 20% lower. While some decarbonisation initiatives might prioritise social responsibility, others prioritise the business line. Future technical and regulatory advancements may alter the profitability of acts, but one thing is for sure: each mine—and each company—will have a different economic case. Mining businesses have traditionally approached sustainability primarily from a local perspective, but major global action will be needed to achieve a 1.5°C to 2.0°C route. Some large mining corporations have established their sustainability committees to indicate that mining is embracing the tide of corporate sustainability reporting and action.

The mining sector is taking more action on climate change because of portfolio reviews of commodities, target-setting, and stakeholder engagement. However, these steps are insufficient to achieve the 1.5°C to 2.0°C scenario. They may not meet societal expectations, as increasingly expressed by investors calling for disclosures, businesses requesting that their suppliers decarbonise, and localities calling for environmental action. Mining businesses may think about more aggressive decarbonization and resilience strategies if they are worried about their long-term reputation, “license to operate,” or contribution to decarbonisation initiatives.

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Chapter 13

Decarbonization in Maritime Sector and Its Path to Sustainability



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1 Introduction

In addition to a large number of ecosystem services that coastal environments provide to humanity, they have a great capacity to store and sequester carbon from the atmosphere (Sapkota and White, 2020). These ecosystems can be of various types, namely, mangroves, wetlands, marshes, seagrasses, etc. The carbon stored in these coastal areas is known as blue carbon (Zeng et al., 2021). Blue carbon is defined as the carbon captured by marine organisms. Ocean sediments are a reservoir of blue carbon of great importance, as it can remain there for up to thousands of years (Kuwae et al., 2022; Macreadie et al., 2019).

However, these systems are seriously threatened of several factors, such as climate change, rising sea levels, overfishing, water pollution, introduction of invasive species, ocean acidification, change in land use, habitat destruction and population growth (Gedan et al., 2009; Lovelock et al., 2017). These ecosystems must be conserved and/or restored as they play a major role in maintaining biodiversity, ecosystem function and local livelihoods (Zeng et al., 2021).

Protection of these vital habitats through blue carbon funding has paled in comparison to the funding given to projects focused on forestry and agriculture. Some blue

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carbon methodologies have been certified by various voluntary carbon markets, for example, those intended to protect habitats and species, those designed to mitigate erosion in coastal areas, those intended to improve water quality, etc. (Wylie et al., 2016). Its implementation, however, is very scarce or in some cases non-existent (Sapkota and White, 2020). Some carbon offset credit schemes are implemented for mangroves and salt marshes, but to date none for seagrass meadows (Kuwae et al., 2022).

Despite the above, the role of blue carbon in coastal ecosystems as a climate mitigation measure has strongly drawn the attention of various politicians and scientists (Macreadie et al., 2019). In recent years, public policies have begun to be formulated for the implementation of compensation mechanisms in coastal projects for the decarbonization of the atmosphere. Approximately 20% of the countries that participated in the 21st Conference of the Parties to the UN Climate Change Convention (UNFCCC) have committed to using coastal ecosystems as a climate change mitigation option. It is estimated that around 2.6 million ha of mangroves (20% of the world's total marine ecosystems) could be financed through carbon finance. If the value of US\$5–9.4/tonnes CO₂ equivalent is taken as the carbon price, at least half of this proportion could be financed for up to 30 years, capturing up to 29.8 million tonnes of CO₂ per year (Zeng et al., 2021).

2 Impact of Mining on Coastal Ecosystem

Heavy metals are natural components of the earth; however, anthropogenic activities together with some meteorological phenomena (exacerbated by climate change) have increased their concentration levels in coastal ecosystems (Venkatramanan et al., 2018). Heavy metals are easily transported from the land to the coast through floods during the rainy season, generating an increase in the concentration levels of heavy metals in different coastal ecosystems (Sundaramanickam et al., 2016). These are deposited in the sediments and remain there for a long time (Wojciechowska et al., 2019), causing toxic effects on the organisms that inhabit these ecosystems.

Almost all mining processes produce waste, which represents one of the main sources of soil, water, air and biota contamination (Younger and Wolkersdorfer, 2004) (Valente et al., 2011). Abandoned mines leave tailings that contain many iron sulphide ores, creating acid mine drainage. This can severely contaminate soils and affect water quality because the low pH makes metals more bioavailable.

The increase in heavy metals in estuaries, lagoons and mangrove environments not only affects their biological and economic productivity but also their ability to sequester carbon from the atmosphere (Abraham et al., 2007). Mining activities carried out in coastal areas (particularly opencast mining) increase transport processes, such as erosion; that is why the sediments in these ecosystems experience a change from sandy to silty texture. Fine particles absorb soluble metals from natural waters and deposit them in sediments (Aurangzed et al., 2014; Sundaramanickam et al., 2016; Pellegrini et al., 2017). This is particularly harmful because plants absorb

these metals from the sediments, causing a toxic effect on them (Kamari et al., 2017). This reduces their productivity and therefore their ability to capture carbon.

Different efforts have been made to reduce the effect of mining on coastal ecosystems. For example, Hernández-Pérez et al. (2021) studied the different species of wetland plants to identify the best candidate for removing metals from sediments and water in polluted coastal areas. According to these authors, the best plant is the one that absorbs the greatest number of metals in the rhizosphere without translocating them to the aerial part. This reduces toxicity because in these plants the metals do not easily reach the leaves where photosynthesis takes place. These techniques not only serve to remedy coastal ecosystems but also help to restore the initial conditions of productivity and thus optimize the capture of carbon from the atmosphere.

3 Sustainable Development of Coastal Ecosystems

Coastal ecosystems are the coastal areas that range from the pluvial basins to the maritime limits of the continental shelves (Singh, 2020). They are delimited mainly by ecological, physical, and biological criteria and that go beyond the political borders between countries. These ecosystems provide goods and services, such as primary production, fishing, habitat for different species, barrier to different meteorological phenomena, energy source, and climate regulation, among others (Sherman et al., 2017).

These ecosystems are of great economic importance because about 80% of fishery production takes place in these areas (Pauly and Lam, 2016). In fact, the average productivity of coastal ecosystems is about three times the productivity of the open ocean. Additionally, 70% of the economically important species are part of or have been developed in this type of ecosystem (Flores Verdugo et al., 2007). The high fertility of coastal lagoons and estuaries maintains a rich and complex food chain, characterized by high fish production. According to the Food and Agriculture Organization of the United Nations (FAO, 2022), global fishery and aquaculture production in 2020 was around 177.8 million tonnes of aquatic animals. The average natural production of fish in coastal lagoons is around 100 kg/ha/year. In fact, many organisms have been shown to do better in estuaries than in coastal waters. This is explained by the high amounts of nutrients and primary productivity in the lagoons, compared to the adjacent ocean. In the case of mangroves, their productivity is higher than that of forests located in rainy areas, and even like that of tropical crops. Proportionally, the productivity of mangroves is twenty times higher than the productivity of open sea and is even five times higher than that of upwelling areas. Unfortunately, marine ecosystems are threatened by climate change, pollution, and government mismanagement, among other causes (Sherman et al., 2017).

In recent years, three summits related to the environment have been held in which the leaders of each nation have committed to carry out various actions aimed at the conservation of coastal ecosystems. The most important agreements are presented below (Meira-Carda, 2019).

- *United Nations Conference on Environment & Development (UNCED)*, Rio de Janeiro, Brazil, 3–14 June 1992, AGENDA 21, Chapter 17: (1) Meet food needs by preventing the degradation of coastal ecosystems and (2) promote the sustainable development of coastal areas.
- *World Summit on Sustainable Development (WSSD)*, Johannesburg, South Africa, 26 August–4 September 2002: (1) Protect marine ecosystems from human activities on land, (2) Promote responsible fishing activities, and (3) Establishment of marine protected areas and restore or fishing populations for sustainable production.
- *United Nations Conference on Sustainable Development (UNCSD)*, Rio de Janeiro, Brazil, 20–22 June 2012: Maintain the biodiversity of coastal ecosystems through their conservation and sustainable use for future generations.

If all these commitments are fulfilled, the contamination and degradation presented by these ecosystems are mitigated. Additionally, indirectly, these actions would increase the capacity of these ecosystems to trap and/or prevent the release of carbon (CO₂ equivalent) into the atmosphere. Therefore, it is imperative that projects aimed at preserving coastal ecosystems activate the different mechanisms available to them to participate in the carbon offset markets.

3.1 Use of Technologies for the Sustainable Development of Aquatic Ecosystems

Aquatic ecosystems are exposed to many threats, putting them in a situation of great vulnerability. The list is long, and among them stand out tsunamis, tropical cyclones, oil spills, and uncontrolled fishing, among others. Emerging technologies are of great help to contain the degradation of this type of environment (Muñoz and Le Bail, 2015). Among them, the following stand out:

- *Satellite observation*: This tool is mainly used to detect polluted ecosystems and wastewater discharge sites. They are also used to identify areas where illegal fishing, deforestation activities and changes in land use are practiced over time. Additionally, it serves to identify eutrophic ecosystems by detecting chlorophylla. On the other hand, in recent years, the Time Series Synthetic Aperture Interferometric Radar (TS-InSAR), together with the global satellite navigation system (GNSS) and the use of unmanned aerial vehicles (UAV), is used to study coastal subsidence because of mining activities in the delta of several rivers (Wang et al., 2022).
- *High frequency coastal radars*: These radars emit radio waves that allow the trajectory and speed of surface ocean currents to be studied. This tool provides high spatial resolution information that allows studying ocean currents, wind and wave height. Through this type of technology, illegal fishing activities, oil spills, and early warnings of tsunamis, among others, can be monitored.

- *Satellite drones*: These are unmanned aerial systems and are controlled from a remote location or programmed to fly over certain areas or regions. These devices are used to observe and monitor the territory when the satellites cannot obtain clear images because of cloud cover. With these devices, very high-resolution satellite images are obtained which study different territories to make an evaluation and control of resources. Drones are widely used to estimate changes in the terrain and the amount of material displaced during a defined period.

Although these tools have been developed mainly for productive activities, the combined use of them is becoming increasingly important in the field of coastal ecosystem preservation.

4 Coastal Projects That Participate in the Carbon Offset Market

Different coastal projects exist throughout the world for the conservation and/or restoration of ecosystems, whose direct or collateral effect is decarbonization in the marine sector. Among them, the following stand out (Lane et al., 2017): (1) Restoration/conservation of coastal wetlands and (2) restoration/conservation of blue carbon ecosystems.

4.1 Restoration/Conservation of Coastal Wetlands

As far as coastal wetlands are concerned, these ecosystems provide ecosystem services, such as carbon sequestration and long-term storage. Carbon sequestration takes place mainly in mangroves, marshes, and seagrasses, among others, being stored as blue carbon (Murray et al., 2011; Sapkota and White, 2020).

These types of wetlands are likely to generate an economic incentive, since the Intergovernmental Panel on Climate Change (IPCC, 2014) has developed clear guidelines for wetlands for the inventory of greenhouse gases (GHG). These can be applied to evaluate carbon stocks and GHG emissions in coastal wetlands. To date, following are the four methodologies in the United States for carbon compensation in wetlands:

- *Restoration of California deltaic and coastal wetlands*

This methodology was approved by the American Carbon Registry (ACR) in 2017. The project comprises rice cultivation in seasonal or tidal wetlands. The development of this methodology had the objective of describing quantification procedures for reducing emissions of GHG through the change of land use from fields of rice crops to wetlands, particularly in the Sacramento-San Joaquin Delta, San Francisco Bay Estuary, and coastal areas of California (Deverel et al., 2017).

This methodology makes it possible to reduce GHG emissions by improving carbon sinks. This is achieved by reducing the oxidation of soil organic carbon and by increasing soil organic carbon storage by restoring wetlands (tidal and non-tidal). This methodology could be applied in those seasonal wetlands where changes in baseline carbon stocks and GHG emissions result mainly from oxidation of organic matter. The GHG not emitted when using this methodology are mainly CO₂, CH₄, and N₂O. By using this methodology, it is calculated that between 290 g/m² and 329 g/m² C-CO₂ per year could be captured.

- *Methodology for coastal wetland creation* (Coastal Protection and Restoration Authority of Louisiana [CPRA], 2014)

This methodology was approved by Verified Carbon Standard (Verra) in 2014 and its field of applicability is in the wetlands that have been reclassified as open water. It focuses on creating new wetlands through substrate management. This methodology quantifies the effect that the creation of new wetlands has on reducing GHG. The scope of this methodology includes two different activities: (1) establishment of the substrate, and (2) establishment of the vegetation, to restore wetlands that have degraded to open water. By using this methodology, it is calculated that between 100 g m⁻² and 200 g m⁻² C-CO₂ per year could be captured.

- *Methodology for tidal wetland and seagrass restoration*

This methodology was approved by Verra in 2015 (Emmer et al., 2015). The most important restoration activities in this project are improvement of water quality, manipulation of water salinity, and proper management of native plants along with the separation of invasive species. This methodology describes the procedures for calculating how many tonnes of GHG can be reduced through the management of tidal wetlands. Management includes actions, such as creation, restoration, and/or management of hydrological conditions, sediment supply, salinity characteristics, water quality, and/or native plant communities. By using this methodology, it is calculated that around 1.46 tonnes/ha² C-CO₂ per year could be captured.

- *Restoration of degraded deltaic wetlands of the Mississippi Delta*

This methodology has as demarcation territory the Mississippi River Delta. The methodology was approved by ACR in 2012 and focuses on comprehensive hydrological management and the planting of trees of different species (Mack et al., 2012). The methodology was developed to quantify changes in reference carbon stocks and GHG emissions. For this, the restoration of wetlands through assisted natural regeneration, seeding, or planting of trees is proposed.

The methodologies developed for coastal wetlands are scarce, especially compared to the great ecological, hydraulic, climatic, and geological variability prevailing between them. That is why these methodologies tend to be very specific for certain regions of the world. Despite the above, these methodologies could be used as a basis to develop others for similar ecosystems in other parts of the world. For example, the methodology developed by Mack et al. (2012) can serve as a basis for other deltaic ecosystems, such as Niger Delta in Niger, Paraná Delta in

Buenos Aires, Argentina, Orinoco Delta in Venezuela, etc. However, other possibilities also exist to develop projects that lack specific methodologies, for example, other methodologies, such as focused on the preservation of forests can be applied in coastal wetland projects; in particular, reducing emissions because of deforestation and forest degradation (REDD+) methodologies are the ones that could be adapted for this purpose. For example, VM0006 Methodology for Carbon Accounting for Mosaic and Landscape-scale REDD Projects, v2.2 developed by Verra. This methodology quantifies the GHG reductions generated in REDD+ projects at mosaic and landscape scales. This methodology is applicable to forests that would be deforested in the absence of conservation projects. The same occurs with coastal ecosystems and the conservation actions are very similar (forest management, afforestation, reforestation, and revegetation activities in accordance with rural economic activities). Despite all of the above, this practice is currently not widely extended.

4.2 Restoration/Conservation of Blue Carbon Ecosystems

Blue carbon is the carbon that is stored naturally in marine ecosystems, such as seagrass beds, salt marshes, and mangroves (Lovelock et al., 2017). Blue carbon refers to organic carbon found in soils beneath plants and in long-lived plant tissues (e.g. wood) (Vanderklift et al., 2019). When plant tissue dies, part of the carbon accumulates in the soil, mixing with the previously accumulated organic carbon. Its importance is in its ability to store a large amount of carbon; for example, a mangrove forest can store up to 10 times more carbon than the same area of a terrestrial forest (Duarte et al., 2013). However, compared to the area occupied by terrestrial ecosystems, blue carbon ecosystems cover a small area globally (approximately 80 million hectares) (Murray et al., 2011).

Among the three main blue carbon ecosystems, mangroves contain the largest amount of blue carbon (1,430 tonnes of CO₂ equivalent per hectare) (Murray et al., 2011). These carbon reserves are vulnerable to erosion, pollution, and/or decomposition; therefore, their degradation causes CO₂ to be released into the atmosphere (Lovelock et al., 2017). This fact is highly alarming because the degradation of this type of ecosystem represents approximately 30% of the CO₂ released into the atmosphere (Houghton, 2003). The degradation of blue carbon ecosystems is estimated to cause annual emissions of up to 1.02 billion tonnes of carbon into the atmosphere (Pendleton et al., 2012). Such is the case that it is estimated that coastal blue carbon ecosystems have lost up to one-third of the global total in recent decades (Pendleton et al., 2012).

Blue carbon ecosystems present different threats to their existence, among which the following stand out: change in land use and its contamination with wastewater and human waste (Kuwaie et al., 2022). Seagrass meadows have been primarily affected by sewage pollution; this generates an enrichment of nutrients (eutrophication), giving rise to the growth of invasive photosynthetic organisms that cloud the water and block the passage of light to the bottom (Le et al., 2019). Furthermore, the wetlands are

intentionally drained by humans for use as a farmland or for tourism development (Gedan et al., 2009). On the other hand, about 40% of the mangroves have been converted into bodies of water destined for aquaculture, tourism, or commercial developments (Alongi, 2002).

That is why, the inclusion of projects for the preservation of marine ecosystems in the carbon offset market is essential to reduce the concentration levels of CO₂ in the atmosphere. Unfortunately, the financial mechanisms and policy frameworks necessary to facilitate investment in preserving these ecosystems are still underdeveloped (Vanderkluft et al., 2019). However, there are opportunities to improve current practices.

Targeted blue carbon ecosystem preservation projects can be financed through the clean development mechanism (CDM) formalized in the Kyoto Protocol (Zhao et al., 2016). However, most of the projects framed in the CDM are for improving energy efficiency; a low percentage are the projects aimed at the preservation of forests and, to date, there is only one blue carbon project (mangrove afforestation in Indonesia) that has received CDM certification. One of the strategies to encourage the financing of projects that produce blue carbon credits is to exalt a great variety of ecosystem services offered by this type of habitat (in addition to CO₂ sequestration). That is, selling them as projects with multiple co-benefits, which include: (1) protection of habitats and species, (2) buffer zones for tropical phenomena, (3) erosion mitigation and tourism development zones, (4) sustainable fishing production areas, (5) improvement of water quality among others (Barbier et al., 2011; Wylie et al., 2016). However, the challenge facing this strategy is to quantify reliably the multiple benefits mentioned in order to attract capital investment.

Although the financing of blue carbon projects has been scarce and presents several challenges, there are some projects in different countries that have been financed through different mechanisms. Among them, the following stand out (Wylie et al., 2016):

- *Project 'Yagasu'*: This project was implemented in Indonesia–North Sumatra by Livelihoods and Yagasu. The project was verified by Verra and it is about storage of carbon in mangrove and wetland ecosystems. The project aims to restore mangroves by planting mangrove species and restoring the original hydraulic systems, recreating an ecosystem, and promoting economic activities, such as fishing for fish, crabs, shrimp, honey, etc. (French Agricultural Research Centre for International Development [CIRAD], 2022). This project was carried out in an approximate area of 5,000 hectares and is valid for 20 years. Through this project, up to 2.1 million tonnes equivalent of CO₂ can be stored for 20 years.
- *Project 'Blue forest Madagascar'*: This Project was implemented in Ambanja and Ambaro Bays, Madagascar, and was developed by 'Blue Ventures' in an approximate area of 26,000 hectares. It was financed by REDD and Plan Vivo. This project, started in 2019, was the first blue carbon project in Madagascar, and aims to conserve and reforest around 1,200 hectares of mangroves. This project has the goal of avoiding the emission of 1,300 tonnes of CO₂ per year (Aigrette et al., 2021).

- *Project 'Markets and mangroves'*: This project was implemented in Mekong Delta, Vietnam and developed by SNV Netherlands, International Union for Conservation of Nature (IUCN). The project was financed by Naturland Organic Shrimp Certification in an approximate area of 1,715 hectares. The project supports the restoration and protection of mangroves while strengthening the livelihoods of small shrimp farmers. The farmers who participated in this project received payments for forest ecosystem services (550 households), with a total amount of almost VN\$900,000,000 (Algemeen nut Beogende Instelling [ANBI], 2023).
- *Project 'Mikoko Pamoja'*: This project was implemented in Gazi Bay, Kenya and developed by Association for Coastal Ecosystem Services (ACES), Kenya Marine Fisheries Institute, Earth Watch Institute, Napier Edinburgh University, and Plan Vivo Foundation. The project area covered 117 hectares and received financing from Voluntary Carbon Credits – Plan Vivo. Its objective is to generate incentives for the protection and restoration of mangroves through the participation of the community. Its main activities are afforestation/reforestation-avoided deforestation. Through this project, around 2,454 tonnes of CO₂ is reduced (Juma et al., 2023).

The aforementioned projects demonstrate that it is possible to obtain financing for the restoration of coastal ecosystems. However, the methodologies for these ecosystems are scattered from one another. If approved, conservation and restoration projects in these areas would be encouraged. This would reduce project implementation costs and make them more sustainable over time.

5 Conclusions

For a long time, coastal ecosystems have remained relatively untouched by the carbon offset market; however, in recent years, their great capacity to sequester GHG is recognized. Although there are few methodologies applied to coastal ecosystems, there is enormous potential to use existing ones and even develop new ones. The conservation of these areas not only allows the capture of GHG but also prevents their degradation from releasing these gases into the atmosphere. From an environmental perspective, taking advantage of marine ecosystems is more sustainable than other types of projects.

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